University of Alberta

Reclamation of Hydrocarbon Contaminated Soils in Central Alberta using Compost, Fertilizer and Native Species

by

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ABSTRACT

A two year field experiment was implemented to evaluate amendment (municipal solid waste and sewage biosolid compost, inorganic fertilizer, compost and fertilizer blend) and seeding (native grass mix) effects on reclamation of diesel fuel and crude oil contaminated soils in central Alberta. Contaminated soils amended with compost or compost-fertilizer had the highest vegetation cover and biomass and the lowest concentration of hydrocarbons at the end of the study. Fertilizer treatments had poor vegetation cover and hydrocarbon concentrations were similar to the unamended treatment. Seeding increased the cover of perennial, native grasses in all amendment treatments, with the greatest cover found in compost and compost-fertilizer amended soils. Total canopy cover and final hydrocarbon concentrations did not differ between seeded and unseeded treatments. Overall, compost appears to be an effective amendment for reclamation of diesel fuel and crude oil contaminated soils and seeding is beneficial for reducing cover of exotic forbs.

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1. RECLAMATION OF HYDROCARBON CONTAMINATED SOILS: AN INTRODUCTION

1.1 RECLAMATION PRACTICES

Approximately 60% of contaminated sites in Canada contain petroleum hydrocarbons (CCME 2001), and in Alberta and Saskatchewan alone there are approximately 100,000 hydrocarbon contaminated sites (McIntyre 2005). Hydrocarbon spills negatively impact the environment due to plant and microbial toxicity, reduction of nutrient availability and impaired hydrologic conditions (Chupakhina and Maslennikov 2004). Petroleum hydrocarbons pose a health risk due to carcinogenic compounds such as benzene and the potential for migration into water sources (ATSDR 1999). As a result, provincial and federal regulations require contaminated lands to be reclaimed to an equivalent land capability (Province of Alberta 1995, Government of Canada 1999, CCME 2001). On disturbed sites reclamation success is evaluated by soil and vegetation quality, however on contaminated sites remediation of the contaminant is also required.

Remediation involves the removal of the contaminant through physical, chemical or biological methods (Riser-Roberts 1998). Physical methods are useful where the contaminant is volatile, and the soil is mechanically tilled to encourage volatilization and displacement of the compounds into the air. Chemical methods are typically energy intensive and involve soil washing processes such as peroxide spraying, chemical extraction and supercritical fluid oxidation or extraction. Biological methods utilize microorganisms to metabolize and ultimately transform contaminants to biomass, carbon dioxide and water (Liu and Suflita 1993, Dua et al. 2002). In the interim, contaminants can be converted to a compound that is either less toxic, less mobile, bound to soil components or incorporated into soil humus in a non-available form. These biological methods commonly require fewer energy inputs, are less expensive and can be conducted on site without soil excavation (Gallego et al. 2001). Bioremediation of contaminated soils is becoming an important alternative to physical and chemical treatment methods due to its efficiency, economics and stewardship (Bartha 1986, Margesin et al. 2000).

At present most remediation practices are conducted separately from revegetation efforts, despite the common end goal of reclamation. Bioremediation can occur on a contaminated site without soil excavation allowing revegetation efforts to commence at the same time (Flathman and Lanza 1998). Unifying these two practices is economically and environmentally favourable as microbial and plant species are similarly affected by environmental contaminants. In addition, microorganisms and vegetation can operate synergistically to remediate contaminated soils. Therefore, techniques to stimulate bioremediation should theoretically benefit vegetation growth and survival.

1.2 Composition of Petroleum Fuels

Due to the processing involved for petroleum fuels, the composition of hydrocarbon compounds and their impact on soils varies greatly amongst fuels. Crude oils are characterized as containing complex, longer chain hydrocarbons such as paraffinic, naphthenic, aromatic and asphaltenic compounds (Westlake et al. 1974). With increased processing, petroleum products contain a greater proportion of saturated compounds compared to aromatic compounds. Song et al. (1990) found gasoline, a low-boiling-point distillate fuel, contained 83% saturates and 15% aromatics, whereas diesel oil, a medium-boiling-point distillate fuel, contained 54% saturates and 45% aromatics.

Degradation of petroleum hydrocarbons in soil is largely dependent on the chemical composition of the compounds. For instance, the persistence of hydrocarbons in the environment increases with molecular complexity, with saturated compounds less persistent than aromatics, and asphaltenes most persistent (Atlas 1981, Geerdink et al. 1996, Demque et al. 1997, Huesemann 1997, Admon et al. 2001). Therefore, the impact of contamination on soil and vegetation properties largely depends on the composition of the contaminant.

1.3 Ecological Impacts of Contamination

1.3.1 Biological Toxicity

Contamination of soils with petroleum products can be toxic to biological life. Chaîneau et al. (1997) observed phytotoxic effects on agronomic plants when grown in soil contaminated with fuel oil (60% saturates, 30% aromatics and 10% resins). Seed germination was reduced in the presence of hydrocarbons in the following decreasing order: *Helianthus annuus* L. (sunflower), *Phaseolus vulgaris* L. (bean), *Triticum aestivum* L. (wheat), *Trifolium repens* L. (clover), *Zea mays* L. (maize), *Hordeum vulgare* L. (barley), *Lactuca sativa* L. (lettuce). The effects of light and heavy fractions of fuel oil were studied on *Zea mays* seed germination, and the light fractions (naphtas) caused the most significant reduction in seed germination relative to the heavy fractions (compounds > C15). Based on phytotoxicity tests, the light fractions were 20 times more toxic than the heavy fractions of fuel oil. Oil can penetrate seeds and alter the metabolic reactions and/or kill the embryo by acute toxicity (Udo and Fayemi 1975), therefore resistance of seeds to oil toxicity relates to the structures of their cell walls (Terje 1984).

Chaîneau et al. (1997) also studied the effect of fuel oil on growth of *Phaseolus vulgaris, Triticum aestivum* and *Zea mays* and found significantly reduced dry biomass of aerial parts for all species in the presence of oil. Adam and Duncan (1999) found reduced plant height and root biomass in numerous grass, forb and commercial crop species grown on diesel fuel contaminated soil (25 and 50 g diesel kg^{-1} soil) relative to uncontaminated soil. In the latter study, plant roots avoided diesel fuel contaminated soil where there was uncontaminated soil to grow into. Amakiri and Onofeghara (1983) found that older *Zea mays* plants (9 to 12 weeks) were more tolerant of crude oil contamination compared to younger plants. The negative effect of hydrocarbons on plant growth can be attributed to inherent toxicity of hydrocarbons but also environmental and soil conditions.

Development of the legume *Vicia sativa* L. (common vetch) was actually stimulated when grown on diesel fuel contaminated soil (Adam and Duncan 2003). Plants grown in

contaminated soil (5 and 10 g diesel kg^{-1} soil) had approximately half the nodules per plant relative to the uncontaminated control; however nodules from control plants were at an earlier stage of development and not able to effectively fix atmospheric N_2. The stimulation of nodule development in contaminated soils may occur due to limited nitrogen availability associated with contamination, which triggered plants to form associations with nitrogen fixing rhizobia to overcome the nutrient stress.

Infiltration of fuel into soil prevents evaporative losses of volatile hydrocarbons, which are toxic to microbial life (Leahy and Colwell 1990, Sikkema et al. 1995). Non-volatile cyclic hydrocarbons in fuel are also toxic to bacterial membranes (Sikkema et al. 1995). Overall, contamination is generally known to reduce soil microbial diversity as the input of hydrocarbons creates a selection pressure for fast-growing hydrocarbon degraders (Atlas et al. 1991, Lawlor et al. 1997, Margesin et al. 2000). This population shift can be beneficial to detoxify the soil, as these organisms are adapted to use hydrocarbons as an energy source and reduce the concentration of the contaminant in the soil environment.

1.3.2 Soil Chemical Properties

Soil nutrient reserves supply plants and microorganisms with essential elements necessary for growth and survival. Due to the carbon content of petroleum fuels, contaminated soils commonly have elevated carbon concentrations (Roy et al. 2003) and microorganisms commonly immobilize inorganic nutrients to assimilate carbon into biomass (Xu and Johnson 1997). As a result, inorganic nutrients, particularly nitrogen and phosphorus, become limited for both microbial and plant growth following contamination. Microorganisms uptake inorganic nutrients and carbon into their cell structures to produce biomass and survive, therefore limited available nutrients would restrict microbial activity (Huesemann 1994). In plants, nitrogen is the mineral element required in greatest quantities for amino acids, proteins and nucleic acids, and phosphorus is important for energy storage and structural integrity of plant cells (Taiz and Zeiger 1998). As a result, inorganic nutrient limitations in soil have a considerable effect on plant growth and survival. Optimal nutrient ratios for hydrocarbon degradation depend

upon concentration of soil contamination, expected biodegradation rate or half-life, amount of soil organic N and rate of mineralization (Huesemann 1994). Literature values for nutrient ratios suitable for biodegradation range between carbon to nitrogen ratios of 10:1 and 200:1 and carbon to phosphorus ratios of 100:1 and 800:1.

Nutrient and metal availability in soil systems is greatly affected by pH, with acidic conditions leading to greater availability but also potential leaching and alkaline conditions leading to more element insolubility (Taiz and Zieger 1998). As a result, changes in soil pH shift the composition of microbial and vegetation communities. Microbiological populations are most diverse and largest in size under near-neutral conditions (Brady and Weil 1999). Heterotrophic bacteria and fungi dominate this range with the population shifting to fungi under acidic conditions (Huesemann 1994). Plants have wide tolerances for pH, however certain plant species are more adapted to grow under highly acidic or alkaline conditions (Brady and Weil 1999).

The heterotrophic bacteria and fungi responsible for hydrocarbon degradation favour neutral soil environments. Atagana et al. (2003) found the optimal pH range for biodegradation of creosote contaminated soil was between 6.5 and 7.0. Degradation of hydrocarbons in acidic soils can increase significantly by adjusting pH towards the neutral range (pH 5.0 to 7.8) (Leahy and Colwell 1990, Gallego et al. 2001, Guerin 2001). Verstraete et al. (1976) observed that the biodegradation rates of gasoline nearly doubled when the pH of an acidic soil was increased from 4.5 to 7.4.

1.3.3 Soil Physical Properties

Extreme water repellency or hydrophobicity has been observed in hydrocarbon contaminated soils (Sawatsky and Li 1997, Roy et al. 2003). The development of hydrophobic conditions following oil contamination is relatively rare and possibly a combination of several factors such as the properties of the oil spilled, soil moisture at time of spill and prolonged exposure to extreme temperature fluctuations after contamination (Roy et al. 2003). Overall, water repellency only becomes problematic at

low soil water contents (Sawatsky and Li 1997), and appears to be more prevalent in surface soils (Roy et al. 2003).

Li et al. (1997) observed decreased yield of *Hordeum vulgare* grown in highly water repellent contaminated soil relative to uncontaminated soil. Despite the reduced yield, there was no difference in root dry mass between contaminated and uncontaminated treatments. This response appeared to be indicative of water stress in the contaminated soil, as vegetation grown under water limiting conditions typically allocates a larger proportion of carbon to its roots (Li et al. 1994). In addition, leachate generation was greater and water content was lower in the contaminated treatment despite the same quantity of water being added to all treatments (Li et al. 1997).

Optimum soil moisture levels for aerobic microorganisms range between 50 and 80% at field capacity and are similar to those for higher plants (Bossert and Bartha 1984, Alexander 1999, Brady and Weil 1999, Atagana et al. 2003). Thus, microbial utilization of hydrocarbons can be limited by reduced or excessive moisture levels (Gallego et al. 2001, Guerin 2001). At low soil moisture microbial growth or survival may decline, but at high moisture levels reduced oxygen concentrations can shift the microbial community to anaerobic populations that are typically less active (Huesemann 1994).

Contaminated soils generally have low organic matter levels due to previous disturbances at the site (Jørgensen et al. 2000). In addition, contamination threatens organic matter production due to reduced plant growth and microbial activity. Soil flora and fauna are responsible for plant residue decomposition in soil and subsequently converting organic nutrients held in the residue to mineral nutrients essential for plant and microbial survival (Brady and Weil 1999). Without plant growth and ultimate senescence, soil organisms would not have the required biomass and energy sources for organic matter cycling.

Soils with high organic matter contents commonly have low bulk densities and therefore greater water holding capacity and aeration. Oxygen is one of the primary factors that determines the rate at which hydrocarbons degrade in soil systems (Jørgensen et al. 2000,

Gallego et al. 2001, Guerin 2001, Vasudevan and Rajaram 2001). The initial steps of aliphatic, cyclic and aromatic hydrocarbon degradation are aerobic and require oxygen as the electron acceptor in their metabolism. Due to the importance of oxygen in initiating the degradation process, it has been identified as a rate-limiting variable in bioremediation of petroleum contaminated soils (Leahy and Colwell 1990).

1.4 Methods to Enhance Reclamation of Contaminated Areas

1.4.1 Amendments and Bioremediation

Inorganic fertilizers are commonly used in hydrocarbon bioremediation to overcome nutrient limitations, stimulate microbial activity and increase degradation rates (Dibble and Bartha 1979, Song et al. 1990, Breedveld and Sparrevik 2000, Brook et al. 2001, Atagana et al. 2003). Demque et al. (1997) observed increased degradation rates of diesel fuel contaminated sand when fertilized with a 21-7-7 (NPK) blend under field conditions. Final hydrocarbon levels were 61% lower in treated soils relative to control soils after 14 weeks. McIntyre (2002) found hydrocarbon levels decreased 46% in fertilized treatments compared to 4% in control treatments after a 3 year period with in situ treatment of diesel fuel contaminated soil. Using a lab-scale bioreactor, Gallego et al. (2001) achieved >90% degradation in diesel fuel contaminated soil with fertilizer application compared to 35% with unfertilized soil. However, Carmichael and Pfaender (1997) and Seklemova et al. (2001) observed little mineralization of phenanthrene and diesel fuel following fertilizer application, and suggested inorganic nutrients stimulated microbial metabolism of readily degradable carbon compounds rather than metabolism of the contaminant. The form of nitrogen in the fertilizer can have a large impact on the rate of hydrocarbon degradation. Brook et al. (2001) investigated the effect of ammonium nitrate, ammonium sulfate, potassium nitrate and urea on degradation of diesel fuel contaminated soils. Increased degradation was observed in soils treated with fertilizers containing ammonium as the primary form of nitrogen. This likely occurred because ammonium is the form of nitrogen available for microbial uptake.

Organic materials that contain sufficient quantities of nutrients can be applied to contaminated soils to enhance soil quality and bioremediation success (Hupe et al. 1996, Namkoong et al. 2002). In addition, certain organic materials have greater microbial diversity than fertile soils, which can further enhance degradation rates (Hupe et al. 1996, Namkoong et al. 2002). However, attention must be paid to the rate at which organic amendments are applied due to the high adsorption capacity of organic materials. High application rates can lead to reduced bioavailability of hydrocarbons to microorganisms, which can affect microbial utilization of the contaminant (Chaîneau et al. 2003).

Application of sewage sludge or compost (food waste and wood chips) to diesel oil spiked soil led to rapid degradation of hydrocarbons (Namkoong et al. 2002). After 30 days in a laboratory setting, soils amended with either sewage or compost (50% application rate based on wet mass of soil) showed 98% degradation. Hupe et al. (1996) had similar results when treating diesel oil impacted soils with kitchen and garden waste compost in test bioreactors. The researchers observed that the cumulative oxygen consumption from degradation of diesel oil in soil increased as compost comprised a higher proportion of the soil/compost mix (11 to 33% based on dry mass of soil). Giles et al. (2001) and Brown et al. (1998) also observed increased hydrocarbon degradation of weathered oil following compost addition to biopiles with 74% removal observed after 24 weeks and 55% degradation after 16 weeks, respectively.

The maturity of composted materials can greatly affect hydrocarbon degradation, as immature composts are characterized by elevated carbon to nitrogen ratios. Addition of readily decomposable carbon materials to contaminated soils can act as a competing nutrient source for microorganisms and ultimately decrease hydrocarbon degradation (Chang et al. 2001, Fernández et al. 2001). Beaudin et al. (1999) found degradation of total mineral oil and grease in contaminated soil increased as the carbon to nitrogen ratio decreased. After 30 days of composting contaminated soil with maple leaves and alfalfa, hydrocarbon degradation was 43% at a carbon to nitrogen ratio of 17 and negligible at a carbon to nitrogen ratio of 40.

Since inorganic fertilizers and organic materials contain nutrients primarily in the rapid and slow release forms, respectively, both amendments are commonly applied to contaminated soils to improve soil quality and stimulate bioremediation. However, the majority of studies that have applied both inorganic and organic amendments (Chang and Weaver 1998, Duncan et al. 1999, Jørgensen et al. 2000, Chaw and Stoklas 2001, Giles et al. 2001, Guerin 2001, Vasudevan and Rajaram 2001) did not investigate the use of the amendments individually and in combination. Without such an investigation, results are difficult to interpret as it is impossible to separate the potential contributions of the organic and inorganic amendments.

1.4.2 Amendments and Revegetation

Due to soil quality restrictions on plant establishment and growth, amendments are commonly applied to contaminated soils to facilitate plant growth. McIntyre (2002) observed vegetation canopy cover was 23% higher on fertilized plots relative to control plots after 3 years of in situ treatment on diesel fuel contaminated soil. In disturbed uncontaminated soils, application of municipal solid waste compost had a linear relationship with increasing plant yield under field (Cuevas et al. 2000) and greenhouse (Iglesias-Jimenez and Alvarez 1993) conditions. Cole et al. (1995) investigated the use of mature yard waste compost and uncontaminated soil in revegetating pesticide contaminated land. Greenhouse pots were planted with *Zea mays* and monitored for 40 days. Total *Zea mays* growth decreased while total weed growth increased in the compost mixes compared to the contaminated and uncontaminated soil mixes. The compost material used in the study contained a high population of viable weed seeds, which may have been attributable to the composting process.

1.4.3 Revegetation and Bioremediation

Bioremediation in association with vegetation is known as rhizosphere bioremediation, phytostimulation or plant-assisted bioremediation (Hue et al. 2002). Establishment of vegetation on hydrocarbon contaminated soils can increase degradation rates due to improved aeration, increased soil organic matter and enhanced microbial activity (Aprill

and Sims 1990). Plant roots exude organic and inorganic substances, which are used by microbial populations as an energy source (Anderson et al. 1993, Reilley et al. 1996, Flathman and Lanza 1998, Palmroth et al. 2005). Bacterial activity can be two to three orders of magnitude higher in the rhizoplane and rhizosphere of the plant compared to the surrounding soil environment, and fungi and actinomycete growth can also increase in the plant zone (Aprill and Sims 1990, Palmroth et al. 2002). Since microbial degradation is the dominant mechanism in bioremediation, stimulation of microbial activity can greatly enhance biodegradation rates. However, to maximize vegetation establishment on contaminated soils, organic and/or inorganic amendments are commonly added to the soil prior to seeding and/or during revegetation.

Aprill and Sims (1990) studied the effect of perennial tall grass prairie species on degradation of four different polycyclic aromatic hydrocarbons (PAH) in soils amended with cow manure and fertilizer. Disappearance of the PAHs in the soil was consistently greater in seeded versus unseeded treatments. A field experiment conducted in Texas tested the effect of seeding annual grass species on fertilized crude oil contaminated soil (Nedunuri et al. 2000). After 21 months, hydrocarbon degradation was enhanced by 40 to 58% on seeded treatments compared to ~25% on control treatments. Reilley et al. (1996) evaluated the effect of perennial forage species on degradation of PAH contaminated soils that were fertilized. After 24 weeks in a greenhouse, vegetated soils had between 30 and 44% greater PAH degradation than unvegetated soils.

Vegetation does not consistently increase rates of hydrocarbon degradation. Bailey and McGill (1999) found no beneficial effect of annual cereal, pulse and forage crops on biodegradation of fertilized oil and creosote contaminated soils. They suggested that the weathered and aged properties of the contamination (oil contamination from 1947 and creosote contamination from 1924 to 1988) could have limited degradation in the rhizosphere of the vegetation. Vouillamoz and Milke (2001) also observed little difference in diesel fuel degradation between seeded (*Lolium perenne* L. (perennial ryegrass)) and unseeded treatments amended with compost (source separated green waste) and fertilizer. After 12 weeks, hydrocarbon degradation in unseeded treatments

was 89% and in seeded treatments degradation increased to 94% (based on 30% compost application rate (wet mass)). The study concluded that compost was more active in stimulating biodegradation than vegetation.

1.5 KNOWLEDGE GAPS

The majority of research conducted in biodegradation, plant-assisted bioremediation and plant suitability for revegetation has been conducted in laboratories or greenhouses. Under such experimental conditions, environmental factors are typically optimized to eliminate 'noise' in the data. These small-scale studies are valuable to understand the mechanisms behind treatment responses (Nedunuri et al. 2000). However, it is unrealistic to assume that these findings will hold under field conditions due to confounding environmental factors such as moisture, temperature, pH and concentrations of inorganic nutrients and metals (Nedunuri et al. 2000, Bengtsson and Zerhouni 2003). Blackburn (1998) found large discrepancies between lab and field based results, with lab values typically exceeding field values by 20 to 100%. Due to the optimization of conditions in small-scale studies, biodegradation rates typically slow as experimental scales increase (Davis et al. 2003). Thus, there is a need for bioremediation and plant suitability research conducted under field conditions.

Revegetation of disturbed and contaminated soils is commonly conducted using agronomic species rather than native species. An understanding of agronomic species tolerant to contamination is important for reclamation projects with an agricultural end landuse. However, many disturbed and contaminated sites exist in natural areas where seeding with agronomic species is not only unsuitable but can be restricted by legislation (Province of Alberta 1995). Thus, there is a need to evaluate native species performance on contaminated lands.

Application of amendments to contaminated soils can accelerate biodegradation of contaminants and establishment of vegetation. However, the source materials for certain organic amendments are 'waste products' such as manure, sewage sludge and municipal

solid waste, which contain metals and organic compounds (Gallardo-Lara and Nogales 1987, He et al. 1992, Villar et al. 1993, He et al. 1995, Cuevas et al. 2000, Zhang et al. 2000, Marcote et al. 2002). In addition, application rates of inorganic or organic amendments in excess of soil/plant requirements have the potential to move off site through leachate. Therefore it is important to study the potential risks associated with amendment application under field conditions.

Successful reclamation of contaminated sites is required to minimize the environmental and health risk associated with the site. However, revegetation and remediation efforts are commonly conducted independently, despite common limitations and the potential for synergy amongst mechanisms. Therefore, it is important to test the effectiveness of combined revegetation and remediation techniques under field conditions.

1.6 Thesis Organization

This thesis is divided into two research chapters followed by a synthesis chapter. In the first research chapter, the impact of contamination on revegetation and the effect of amendments and seeding on revegetation success are discussed. In the second research chapter, the effect of amendments and seeding on bioremediation and the relative persistence of different hydrocarbon fractions over time are examined. In the synthesis chapter, an overview of the main research findings is provided and future research directions and management implications are discussed.

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2. REVEGETATION OF HYDROCARBON CONTAMINATED SOILS

2.1 INTRODUCTION

Hydrocarbons are the most common environmental contaminants in industrialized countries (Palmroth et al. 2002). For instance, there are over 28,000 wellsites and 18,000 pipelines decommissioned and an additional 15,000 new wellsites drilled each year in the province of Alberta (Powter 2003). In addition to vegetation stripping and soil disturbance, a portion of these sites will be contaminated with petroleum hydrocarbons due to mechanical failures, leaks or spills (Vasudevan and Rajaram 2001, Chaîneau et al. 2003, Juteau et al. 2003). Due to environmental and health concerns, contaminated sites must be reclaimed to equivalent land capability (Province of Alberta 1995, Government of Canada 1999). However, contaminated soils are typically low in organic matter and nutrients, elevated in contaminant concentrations and lack desired seed sources, which can limit reclamation success (Atlas 1981, Leahy and Colwell 1990, Huesemann 1994).

Application of organic and/or inorganic amendments to contaminated soils can ameliorate poor soil conditions that may be limiting vegetation establishment (Stukenholtz et al. 2002). Organic amendments are typically high in organic matter, have a neutral pH and contain nutrients in a slow-release, organic form (Levi-Minzi et al. 1990, Iglesias-Jimenez and Alvarez 1993, Sikora and Yakovchenko 1996), whereas inorganic fertilizers supply readily available inorganic nutrients to the soil and vegetation. Municipal solid waste compost is readily available in municipalities with urban composting programs. Such composting operations, process waste materials that would otherwise be sent to a landfill and convert them into a useful organic amendment. However, due to the heterogeneous nature of the waste products, the compost material can contain elevated concentrations of undesirable compounds such as metals and salts (He et al. 1992, Villar et al. 1993, He et al. 1995). Inorganic fertilizer application can lead to nutrient leaching and acidification if applied at heavy rates (Brady and Weil 1999).

Establishment of vegetation on disturbed sites is required for successful reclamation to stabilize the soil surface, reduce erosion and contribute to nutrient cycling and soil aeration (Schnoor et al. 1995). However, absence of adequate seed sources can limit establishment of desired plant communities on disturbed or contaminated sites. Seeding species that are tolerant of environmental conditions at the disturbed site is essential to improve revegetation success. Revegetation research on disturbed sites is commonly conducted with agronomic species that are not native to the geographical area (Huang et al. 2004).

Numerous plant suitability studies have been conducted under controlled greenhouse conditions, with environmental factors optimized. However, field variables such as precipitation, temperature and soil structure can greatly affect vegetation success. Therefore, a field experiment was conducted to investigate revegetation success on contaminated soils.

2.2 Research Objectives

- Investigate the effect of diesel fuel and light crude oil contamination on revegetation of disturbed soils over two growing seasons
- Determine the effect of amendments and seeding on revegetation of contaminated soils over two growing seasons

2.3 Methods

2.3.1 Study Area

The field experiment was established at the City of Edmonton Waste Management Centre in Edmonton, Alberta. The experiment was conducted from late June 2003 to late August 2004. The area has a mean annual temperature of 3.1 °C, the mean monthly temperature varies between -16.6 and 22.2 °C and the mean annual precipitation is 466 mm (Environment Canada 2004). At the research site, temperature conditions over the study

period were similar to climate normals for the area, however rainfall was generally below normal in both growing seasons with the exception of August 2003 and July 2004 when above average rainfall occurred (Table A.1). The soil on site was an industrially disturbed black chernozem with a sandy loam texture. Due to the disturbed nature of the soil, total organic carbon levels were low relative to black soils in the region (van Ryswyk et al. 1965, Anderson and Coleman 1985, McGill et al. 1988) and average background soil hydrocarbon levels were < 100 mg kg^{-1} (Table 2.1).

2.3.2 Experimental Design

To ensure experimental control, experimental units in the study consisted of 20 L (28.5 cm diameter, 40 cm depth) plastic pails with an internal drainage system (perforated PVC tube (1.9 cm diameter) and sterilized sand (5 cm depth)) to remove leachate periodically (Figure A.1). A randomized complete block design was used in the experiment with three factors applied within each of four replicate blocks (Figure A.2). The factors tested in the experiment were fuel type, amendment combination and seed mix. Soil used in the experiment was a mixture of approximately 40% topsoil and 60% subsoil (based on dry soil mass) to represent an admixed disturbed soil. Soils were artificially contaminated with fuel to ensure experimental control over the type and uniformity of contamination. The fuel and amendment combinations were incorporated into the soil using a cement mixer (one treatment replicate mixed at a time). Soils were settled into the pails by gently tapping the bottom of the pail on the ground and then pails were buried in the ground (to an approximate depth of 35 cm) to ensure the treatments would be exposed to ambient environmental conditions over the study period. Pails were seeded with the specified seed mix after soil placement and the surface was lightly raked to a depth of 0.5 cm to enable incorporation and minimize loss of seed.

Soils were contaminated with either diesel fuel or light crude oil, as they represent common contaminants in soil systems; an uncontaminated control was also monitored. Diesel fuel is characterized as containing predominantly light hydrocarbon fractions and light crude oil is known to contain a mixture of light and heavy fractions. Diesel fuel was

obtained from Hughes gasoline station and crude oil was supplied by the ESSO Strathcona refinery in Edmonton, Alberta. Fuel was applied to the soil at a rate of 1% based on dry soil mass (equivalent to 10,000 mg kg^{-1} total extractable hydrocarbons) as this was viewed as a moderate contamination level suitable for bioremediation.

Soils were amended with non-source separated municipal solid waste (MSW) and sewage biosolids compost from the City of Edmonton, ammonium sulphate (21-0-0-24) fertilizer and a compost-fertilizer blend; an unamended control was also monitored. To maintain equal volumes of soil mixture in each pail, the fertilizer and unamended pails contained 23.2 kg of soil (dry mass) and the compost and compost-fertilizer pails contained 14.9 kg of soil (dry mass). The mature compost from the City of Edmonton contained macro and micronutrients, had a neutral pH and was pathogen free (fecal coliforms and *Salmonella*). However, due to the feedstocks of the material, it contained elevated concentrations of salts and certain metals (Cd, Cu, Pb, Zn) (Table A.2). Thus, the material is a class B compost according to Canadian Council of Ministers of the Environment (CCME) guidelines for compost quality (CCME 1995). The compost material was mixed with the soil at a rate of 20% based on dry soil mass (equivalent to 10% based on wet soil volume) in both the compost and compost-fertilizer treatments. This application rate was selected as it increased the total organic carbon level in the treatment soil to background levels for the area. Fertilizer application was aimed at optimizing nutrient levels for microbial activity to ultimately stimulate biodegradation of hydrocarbons (Chapter 3). A C:N ratio of 10:1 was considered ideal for microbial populations (Huesemann 1994), therefore ammonium sulphate was incorporated into the soil at a rate to achieve a hydrocarbon-C:N ratio of ~10:1 (Table A.3). Due to the lower quantity of soil in the compost-fertilizer treatment, 60 g of fertilizer was added to the compost-fertilizer treatment compared to 96 g added to the fertilizer treatment. The resulting C:N ratios in the fertilizer and compost-fertilizer treatments were 11.9 and 11.1 in the diesel soil, 11.1 and 11.8 in the crude soil and 8.6 and 12.1 in the control soil, respectively.

Soils were seeded with either a native grass mix or a native grass and legume mix; an unseeded control was also monitored (Table A.4). The native grass mix included

Agropyron trachycaulum (Link) Malte (slender wheatgrass), *Elymus innovatus* Beal (hairy wildrye), *Deschampsia caespitosa* (L.) Beauv. (tufted hairgrass) and *Calamagrostis canadensis* (Michx.) Beauv. (bluejoint). The native grass and legume mix included the above four species plus *Vicia americana* Muhl. (American vetch) and *Astragalus canadensis* L. (Canada milk vetch). Germination tests were conducted prior to seeding to confirm the viability of the seeds. Native species were selected based on seed availability, fibrous root systems and relative tolerance to salt, hydrocarbons and drought. The seeds were applied to the treatment pails at a rate of 300 pure live seeds m^{-2}, as this is a common rate used in reclamation.

Throughout the 14 month study period, very poor establishment of the legume species in the native grass and legume seed mix was observed (trace cover in 19% and 4% of pails in 2003 and 2004, respectively). Legumes, in particular *Vicia* and *Astragalus* species, are known to be difficult to establish due to physical dormancy from impermeable seed coats (Baskin and Baskin 1998, Raven et al. 1999). Water uptake and gas exchange are the primary limitations to germination for seeds with thick seed coats. Germination of *Astragalus canadensis* occurred over a 5 year period when seeds were sown on soil in a non-temperature controlled greenhouse in Kentucky (Baskin and Baskin 1998). Therefore, seed viability is high for species with physical dormancy but germination can occur over several years after seeding. Due to the unsuccessful growth of the legumes, the native grass and legume pails were excluded from soil sampling in 2004 and no results are presented for this treatment.

2.3.3 Sampling

Soil samples were collected at the beginning and end of the study in June 2003 and August 2004, respectively. Samples were analyzed for total extractable hydrocarbons (TEH) between C_{11} and C_{60} (EPA 3550/8000-GC-FID), trace metals (EPA 6020), fecal coliform and *Salmonella* (APHA 2001), total organic carbon (TOC) (Nelson and Sommers 1996), total kjeldahl nitrogen (TKN) (McGill and Figueiredo 1993), ammonium (NH_4^+) (Maynard and Kalra 1993), nitrate (NO_3^-) (APHA 2001), sulphate

(2004 only) (Combs et al. 1998), pH (Janzen 1993), electrical conductivity (EC) (Janzen 1993), sodium adsorption ratio (SAR) (Janzen 1993) and bulk density (2004 only) (ASTM 2004) (Table A.5). Individual carbon lengths were summed from the TEH analysis into three groups to correspond with fraction 2 (F2) (>C10 to C16), fraction 3 (F3) (>C16 to C34) and fraction 4 (F4) (>C34) hydrocarbons (CCME 2001). In addition, three composite samples of topsoil, subsoil and compost were collected to characterize the initial materials in 2003. Samples were analyzed for above parameters in addition to bulk density and particle size distribution (Sheldrick and Wang 1993) (excluding compost samples). All samples were stored in coolers (~5 °C) during sampling and sent to EnviroTest Laboratories in Edmonton, Alberta for analysis immediately following collection.

In June 2003, one sample was taken from the cement mixer after the soil-fuel-amendment mixture for each treatment replicate had been homogenized. In August 2004, soil samples were collected from two depths in each pail, 0 to 10 cm and 10 to 20 cm, to evaluate differences in the soil profile over the study period. To collect the samples, each pail was removed from the ground, tipped upside down and then cut in half vertically to expose the center of the soil profile. Profile sampling was chosen as the soil was quite rocky and therefore soil coring was not effective. In addition, profile sampling enabled half the pail to be sampled for chemical analysis and the other half to be saved for root biomass collection. The cut edge was lightly scraped to remove potential contamination during cutting and samples were collected using a trowel from the two depths. All sampling equipment was sterilized periodically using a dilute bleach mixture to eliminate cross contamination between treatments. Upon analysis of the 2004 soils data, no statistically or biologically significant differences between the two sampling depths were found for all soil chemical parameters analyzed. Therefore, the depths were averaged for all parameters and the results presented reflect data from 0 to 20 cm.

Following rainfall events in 2003 and 2004, leachate was pumped from each treatment pail using a peristaltic or manual pump. Leachate volumes were recorded for each treatment pail, however no chemical analyses were conducted on the liquid. Removal of

leachate from the pails, ensured soils were at or below field capacity over the study period and were not artificially exposed to anaerobic conditions.

Vegetation assessments were conducted three times over the study period (September 2003, May and August 2004) to capture early and late season species. Total canopy cover (%) and species composition were determined at each time. To evaluate vegetation composition, individual plant species were divided into six functional groups based on life form, origin and growth habit (Table 2.2). The majority of the vegetation surveyed in the study was classified either as annual-exotic-forbs, perennial-exotic-forbs or perennial-native-grasses; therefore the remaining functional groups (annual-native-forbs, perennial-native-forbs, annual-exotic-grasses) were omitted from statistical analysis.

Destructive vegetation sampling occurred at the end of the experiment in August 2004. Aboveground biomass (live vegetation and litter) was clipped from the entire pail area (0.064 m^{-2}) prior to soil sampling and belowground biomass was collected via root washing. To obtain root biomass samples, an intact half pail of soil was placed on a 0.25 cm^2 wire screen and gently shaken. The soil was then sprayed with water to separate the soil particles from the root material. To ensure all the soil and organic material had been removed from the roots, the roots were gently massaged under water until only root material remained. All above and belowground biomass samples were oven dried at 80 ^{o}C for 48 hours then weighed. Both above and belowground biomass data were expressed on a mass per area basis (g m^{-2}) where the total rooting depth was 25 cm.

2.3.4 Data Analyses

All data were analyzed using analysis of variance (ANOVA) procedures (Zar 1999). Normality tests were conducted on the data and the majority of the parameters had normal distributions within a treatment type and the remaining data deviated only slightly from a normal distribution. To determine the effect of contamination on revegetation a one-way ANOVA with blocking was conducted using only the control (unamended and unseeded) treatments within each fuel type. The effect of amendments and seeding on

revegetation was determined separately for each fuel type using a two-way ANOVA with amendment and seeding as main factors. To determine whether treatment responses differed between uncontaminated and contaminated soils, a three-way ANOVA was conducted with fuel, amendment and seeding as factors. Since a comparison between diesel and crude soils was not the objective of the study, separate ANOVAs were conducted for uncontaminated versus diesel soil and uncontaminated versus crude soil. Interactions between factors were tested and multiple comparisons were conducted on each ANOVA procedure mentioned above using Fisher's least significant difference (LSD) test (Zar 1999). Paired t-tests were used to determine differences in vegetation cover between 2003 and 2004. All statistical analyses were conducted using SYSTAT 10 (Systat Software Inc., Richmond, CA) software and a p-value of 0.05 was selected to determine significance.

2.4 Results and Discussion

2.4.1 Vegetation

Vegetation cover and biomass (above and belowground) was affected mostly by contamination and amendments rather than seeding (Figures 2.1, 2.2). In uncontaminated soils, vegetation cover was significantly lower and biomass was lower in the fertilizer treatment relative to the control, compost and compost-fertilizer treatments. In the diesel and crude soils, cover and biomass were significantly lower in the control and fertilizer treatments compared to the compost and compost-fertilizer treatments. Between the two compost treatments, the compost-fertilizer treatment had greater aboveground biomass in contaminated (significant) and uncontaminated soils and greater cover in the crude (significant) and diesel soils.

Seeding had an effect on above and belowground biomass in only the compost treatment (Figure 2.2). Aboveground biomass was significantly greater in the uncontaminated soil and both above and belowground biomass was significantly greater in the crude soil

when the compost treatment was seeded. No differences between seeded and unseeded treatments were observed in the diesel soil.

2.4.2 Contamination Effects

Diesel or crude contaminated soils had significantly greater TEH concentrations in both 2003 and 2004 relative to the control soil (Table 2.3). Due to the properties of diesel fuel and crude oil, the contaminated soils contained different concentrations of hydrocarbon fractions. Over the study, the diesel soil had significantly greater F2 and F3 hydrocarbons than the control soil and the crude soil had significantly greater F2, F3 and F4 hydrocarbons than the control soils. To contrast the fuels, diesel had a greater proportion of F2 hydrocarbons and a lower proportion of F4 hydrocarbons than the crude soil and the two fuels had similar concentrations of F3 hydrocarbons.

Vegetation cover and biomass was significantly lower in diesel and crude soils relative to control soils (Table 2.3), which is consistent with the results of other studies (Adam and Duncan 1999, Bailey and McGill 1999, McIntyre 2002, Robson et al. 2002). Contamination of soils with petroleum hydrocarbons can affect vegetation growth due to an imbalance in nutrient ratios, water retention restrictions and/or phytotoxic effects from compounds in the fuel (Li et al. 1997, Adam and Duncan 1999, Roy et al. 2003). The contamination event increased TOC concentrations in the contaminated soils relative to the control soil (Table 2.3), however the resulting C:N ratio did not rise to a level that would cause nutrient immobilization (Brady and Weil 1999). Water repellency is more common in aged contaminated soils and low soil moisture plays an important role in the severity of the repellency (Roy et al. 2003). In this study, total leachate volumes were ~3 times greater in the contaminated soils compared to the control soil (Table 2.3). Due to the age of the contamination, the leachate results can likely be attributed to the low vegetation biomass in contaminated soils, however, water repellency cannot be ruled out. Phytotoxicity from petroleum contamination depends upon the type and composition of contamination, with lighter hydrocarbon fractions (F2) representing greater toxicity to plant establishment (Chaîneau et al. 1997). Due to the elevated concentration of F2

hydrocarbons in both the diesel and crude soils (Table 2.3), it appears the low vegetation growth on the contaminated soils is most affected by the phytotoxic effects of the fuel.

2.4.3 Amendment Effects

Fertilizer application resulted in significantly greater NH_4^+ concentrations in the fertilizer and compost-fertilizer treatments relative to the compost and control treatments in both the contaminated and uncontaminated soils (Table 2.4). Under aerobic conditions, nitrification converts NH_4^+ into NO_3^- and H^+ ions are released in the process. The accumulation of H^+ ions in the soil has been known to lower soil pH and has been observed following ammonium sulphate application (Brady and Weil 1999). In this study, the pH of the fertilizer treatment significantly decreased over the study period and was ~2 units lower than the other treatments (Table 2.4). Most studies have found vegetation cover and biomass to increase following fertilizer application due to the stimulatory effect of inorganic nutrients (McIntyre 2002, Sellers et al. 2002). The contradictory findings in this study can likely be linked to the acidic pH of the fertilizer treatment (Taiz and Zeiger 1998). On the other hand, fertilizer application with compost did not result in reductions of pH (Table 2.4), which suggests the compost buffered the effects of nitrification (Giles et al. 2001). In addition, the higher initial concentration of NH_4^+ in the compost-fertilizer treatment (Table 2.4) could explain the increased vegetation production relative to the compost treatment.

Due to the source materials of the compost used in the study, the compost contained elevated EC, SAR, Na and certain metals (Cd, Cu, Pb and Zn) (Table A.2). As a result, the compost and compost-fertilizer treatments had significantly greater levels of the above parameters compared to the control treatment (Table 2.4). Overall, EC, SAR, Cd, Cu, Pb and Zn in both compost treatments exceeded CCME guidelines for soil quality based on certain end land uses (Table A.6). However, vegetation growth in both compost treatments did not show signs of developmental damage due to salinity or metal exposure. Vegetation cover was also consistently greater in the compost-amended treatments in contaminated soils and had cover equivalent to the control treatment in

uncontaminated soil. In addition, the change in metal concentrations between 2003 and 2004 was not statistically different between amendment treatments, suggesting the metals in the compost treatments were not bioavailable or mobile. Numerous studies support this hypothesis and have shown that under neutral to alkaline pH conditions, trace metal bioavailability is restricted (Dick and McCoy 1993, Moreno et al. 1996, Stratton et al. 1998, McBride et al. 2000). In another study, Kaschl et al. (2002) found that trace metals in municipal solid waste compost were present in very low percentages (<2.5%) in the bioavailable water extractable fraction. Despite the acidic pH of the fertilizer treatment at the end of the study, significant metal losses were not observed from the treatment (Table 2.4). This observation is likely attributed to the low background concentrations of metals in the fertilizer treatment.

Over the study period, the compost and compost-fertilizer treatments lost significantly more sodium than the fertilizer or control treatments (Table 2.4). Salts in soil systems are very soluble and therefore mobile (Brady and Weil 1999), as a result the dominant mechanism for sodium loss in the compost treatments is most likely through downward water movement or leachate. Thus, consequences on downstream water and soil quality are a potential issue if compost is applied in groundwater recharge areas or on lands with a shallow water table.

2.4.4 Contamination and Amendment Effects

Compost addition to uncontaminated and contaminated soils significantly increased TOC concentrations relative to the fertilized and control treatments (Table 2.4). Levels of TOC in the compost treatments were representative of background soils in the black zone compared to the unamended soil (van Ryswyk et al. 1965, Anderson and Coleman 1985, McGill et al. 1988). Compost addition was also found to significantly increase TKN and NO_3^- concentrations in the compost and compost-fertilizer treatments compared to the other treatments (Table 2.4). Since plants uptake nitrogen in the form of NO_3^-, the positive vegetation growth observed in compost treatments may be partially linked to nutrient availability. Addition of organic matter to the soil in the form of compost

resulted in lower bulk densities in the compost and compost-fertilizer treatments compared to the other treatments in the uncontaminated and contaminated soils (significant) (Table 2.4). However, the sandy loam soil used in the study had a low bulk density, therefore the difference in bulk densities between the treatments has limited biological significance (Li. et al. 2002).

Leachate quantities were lower in the diesel (significant) and crude soils that were amended with compost or compost-fertilizer relative to fertilizer or unamended treatments (Figure 2.3). The reduced leachate in compost treatments can be linked to the increased vegetation biomass in those treatments, but also to increased water retention. Compost increases the organic matter content of soils, which improves water holding capacity and reduces the impact of petroleum induced water repellency (Aprill and Sims 1990). In addition, hydrocarbon compounds in contaminated soils bind to organic matter particles, thereby reducing their bioavailability and phytotoxic effects (De Jonge et al. 1997). Numerous studies have documented increased vegetation productivity following application of mature compost (Cole et al. 1995, Vouillamoz and Milke 2001). Therefore, the combined effects of enhanced nutrient availability, reduced water repellency and decreased phytotoxicity are likely responsible for the significant increase in vegetation growth observed in the diesel and crude contaminated soils.

Vegetation biomass, particularly belowground biomass is known to affect water retention in soils and therefore the production of leachate (Shimp et al. 1993). In this study, leachate from diesel soils was strongly linked to belowground biomass, where leachate volumes decreased in treatments with higher biomass (Figures 2.2, 2.3). However, in the crude soils the relationship between belowground biomass and leachate was less clear. Despite differences in belowground biomass between the compost and compost-fertilizer treatments and the fertilizer and unamended treatments, leachate volumes were not distinctly different (Figures 2.2, 2.3). The hydrophobic nature of crude oil contaminated soils could provide a possible explanation for this observation.

Overall, the compost and compost-fertilizer treatments in the contaminated soil had similar vegetation cover and biomass compared to the same treatments in the uncontaminated soil. On the other hand, the fertilizer and control treatments in the contaminated soils had lower vegetation cover and biomass compared to same treatments in the uncontaminated soils. Therefore, compost was able to ameliorate poor soil conditions in the contaminated soil and support vegetation growth similar to uncontaminated soils.

2.4.5 Contamination, Amendment and Seeding Effects

Seeding with native grasses had no effect on soil properties or total canopy cover, however vegetation composition was different between seeded and unseeded treatments (Figure 2.4). In the uncontaminated soil, seeding increased the cover of perennial-native-grasses in all amendment treatments, with a significant increase observed in the compost-fertilizer treatment. Seeding also significantly decreased the cover of perennial-exotic-forbs in the control treatment and decreased the cover of annual-exotic-forbs in the compost and compost-fertilizer treatments in the uncontaminated soil. In the diesel soil, seeding significantly increased cover of perennial-native-grasses in the compost and compost-fertilizer treatments, which resulted in decreased cover of annual and perennial-exotic-forbs. Similarly, in the crude soil seeding significantly increased the cover of perennial-native-grasses in the compost and compost-fertilizer treatments, which resulted in a decrease of annual-exotic-forbs. Between the compost and compost-fertilizer treatments, there was no difference in perennial-native-grass cover in uncontaminated and contaminated soils. Therefore, addition of ammonium sulphate to compost did not stimulate perennial-native-grass growth. Of the species seeded, *Agropyron trachycaulum* established most successfully and represented the majority of perennial-native-grass cover in most treatments (Tables A.7, A.8, A.9).

Despite the lack of difference in total vegetation cover between unseeded and seeded treatments, the higher proportion of perennial-native-grasses in the seeded treatments represents a considerable management benefit. A number of the annual and perennial-

exotic-forbs that established on the uncontaminated and contaminated soils are listed as noxious or nuisance weeds (Table 2.2), and therefore must be controlled on sites through mechanical, chemical or biological measures (Province of Alberta 2002).

2.4.6 Time

An observation of vegetation compositional changes over time provides insight into future growth patterns of a plant community (Table A.10). In this study, vegetation cover increased for all treatments in the uncontaminated and contaminated soils between 2003 and 2004, with the exception of the control where marginal decreases were observed (data not shown). The composition of the control shifted from primarily annual-exotic-forbs in 2003 to mostly perennial-exotic-forbs in 2004, with a significant increase in perennial-native-grass cover observed in the seeded treatment. In unseeded fertilizer, compost and compost-fertilizer treatments in uncontaminated soil, increased cover in 2004 comprised mainly of annual-exotic-forbs in the fertilizer and compost-fertilizer treatments and both annual and perennial-exotic-forbs in the compost treatment. In the unseeded compost and compost-fertilizer treatments in contaminated soil, the increased cover in 2004 could be largely attributed to annual and/or perennial-exotic forb growth in the diesel soils and to increased annual-exotic-forb growth in the crude soil. In seeded treatments, the increased cover in 2004 comprised a greater proportion of perennial-native-grasses than exotic-forbs in the all soils. Although difficult to forecast, the increased cover of perennial-native-grasses in 2004 in the compost seeded treatments suggests perennial-native-grass species would persist and continue to compete with exotic-forb species.

2.5 Conclusions

- Contamination with diesel fuel or light crude oil significantly decreased vegetation production relative to control soils likely due to phytotoxic effects of the fuel and/or water repellency.

- Fertilizer application decreased soil pH and resulted in decreased plant growth in the uncontaminated soil relative to the control and very low plant growth in the contaminated soils.
- Compost application significantly increased plant growth on diesel and crude soils and had similar growth to the control treatment in the uncontaminated soil. The organic matter in the compost likely enhanced vegetation by reducing the phytotoxic and water repellent effects of the contamination and through the input of available nitrate.
- Seeding with native grasses did not affect total canopy cover, however cover of perennial-native-grasses was greater in all seeded treatments. Thus, perennial native grasses were not present in the seed bank or seed rain, therefore seeding contaminated sites can improve cover of desired species.
- Compost amended (compost and compost-fertilizer) treatments had the greatest cover of perennial-native-grasses, which is likely due to increased organic matter, improved water retention and reduced phytotoxicity (in contaminated soils). However, there was no statistical difference in perennial-native-grass cover between the compost and compost-fertilizer treatments, suggesting that increased ammonium and sulphate concentrations do not stimulate plant growth.
- Successful plant growth in compost amended treatments, suggests elevated EC, SAR and metals did not cause harm to vegetation establishment or survival. However, potential downstream water and soil quality issues may arise following compost application due to sodium loss and should be studied further

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Table 2.1. Mean baseline compost and soil treatment characterization

	Compost[1]	Soil[2]	Soil + Fertilizer[2]	Soil + Compost[2]	Soil + Compost + Fertilizer[2]
Sand (%)[1]	-	73 (0.5)			
Silt (%)[1]	-	14 (0.9)			
Clay (%)[1]	-	13 (0.5)			
Bulk Density ($Mg\ m^{-3}$)[1]	0.53 (0.01)	1.88 (0.02)	1.88 (0.02)	1.74 (0.01)	1.74 (0.01)
TEH ($mg\ kg^{-1}$)	9033 (491)	76 (15)	55 (13)	2053 (292)	2058 (163)
TOC (%)	19.9 (0.6)	1.8 (0.1)	1.6 (0.2)	6.4 (0.3)	6.3 (0.1)
TKN (%)	1.85 (0.01)	0.13 (0.01)	0.2 (0.01)	0.52 (0.02)	0.54 (0.01)
C:N	11.2 (0.4)	14.5 (0.5)	8.6 (0.7)	13 (0.2)	12.1 (0.3)
NH_4^+ ($mg\ kg^{-1}$)	7.5 (0.2)	6.1 (0.3)	811.8 (0.2)	10.7 (1.5)	638.2 (22.9)
NO_3^- ($mg\ kg^{-1}$)	1867 (75)	8.5 (0.4)	8.3 (0.5)	344.4 (19.9)	317.5 (28.8)
pH	7.7 (0.03)	6.63 (0.02)	6.65 (0.04)	7.13 (0.04)	7.19 (0.02)
EC ($dS\ m^{-1}$)	16.7 (0.47)	0.49 (0.02)	0.51 (0.02)	10.18 (0.28)	9.89 (0.11)
SAR	15.9 (0.4)	0.29 (0.01)	0.29 (0.01)	9.69 (0.25)	9.5 (0.12)
Na ($mg\ L^{-1}$)	2387 (89)	11.3 (0.4)	11 (0.4)	1399.2 (40.1)	1355 (22.5)
Cd ($mg\ kg^{-1}$)	3.0 (0.1)	0.5 (0)	0.5 (0)	1.01 (0.09)	1.04 (0.06)
Cu ($mg\ kg^{-1}$)	200 (2.8)	12.8 (0.6)	11 (0.6)	62.6 (5.5)	63.1 (3)
Pb ($mg\ kg^{-1}$)	152 (2.9)	11.1 (0.7)	9.7 (0.5)	47.3 (4.2)	46.5 (2.6)
Zn ($mg\ kg^{-1}$)	646 (9.9)	54.2 (2.6)	47.9 (1.9)	204.8 (17.5)	212.3 (12.9)
Fecal Coliforms (MPNU g^{-1})	<3 (0)	<3 (0)	<3 (0)	<3 (0)	<3 (0)
Salmonella	Not isolated	Not isolated	Not isolated	Not isolated	Not isolated

[1] n = 3

[2] n = 12 (seeding treatments combined)

Numbers in brackets indicate standard error of the mean

TEH, total extractable hydrocarbons; TOC, total organic carbon; TKN, total kjeldahl nitrogen, C:N, carbon to nitrogen ratio; NH_4^+, ammonium; NO_3^-, nitrate; EC, electrical conductivity; SAR, sodium adsorption ratio; Na, sodium; Cd, cadmium; Cu, copper; Pb, lead; Zn, zinc

Table 2.2. Plant species grouped by functional type

Functional Group	Common Name	Scientific Name	Weed Status[1]
Annual Exotic Forb	green foxtail	*Setaria viridis* (L.) Beauv.	Nuisance
	redroot pigweed	*Amaranthus retroflexus* L.	Nuisance
	wild buckwheat	*Polygonum convolvulus* L.	Nuisance
	lamb's quarters	*Chenopodium album* L.	
	kochia	*Kochia scoparia* (L.) Schrad.	
	annual hawk's beard	*Crepis tectorum* L.	Nuisance
	shepherd's purse	*Capsella bursa-pastoris* (L.) Medic	Nuisance
	flixweed	*Descurainia sophia* (L.) Webb	Nuisance
	purslane	*Portulaca oleracea* L.	
	prostrate knotweed	*Polygonum aviculare* L.	
Perennial Exotic Forb	absinth	*Artemisia absinthium* L.	
	perennial sow thistle	*Sonchus arvensis* L.	Noxious
	dandelion	*Taraxacum officinale* Weber	Nuisance
	broad-leaved plantain	*Plantago major* L.	
Annual Native Forb	pygmy flower	*Androsace septentrionalis* L.	
	pineapple weed	*Matricaria matricarioides* (Less.) Porter	
	Canada fleabane	*Erigeron canadensis* L.	
Perennial Native Forb	rough cinquefoil	*Potentilla norvegica* L.	Nuisance
Annual Exotic Grass	canola	*Brassica rapa* L.	
	wheat	*Triticum aestivum* L.	
	barnyard grass	*Echinochloa crusgalli* (L.) Beauv.	
Perennial Native Grass	foxtail barley	*Hordeum jubatum* L.	
	hairy wildrye	*Elymus innovatus* Beal	
	slender wheatgrass	*Agropyron trachycaulum* (Link) Malte	
	tufted hairgrass	*Deschampsia caespitosa* (L.) Beauv.	
	bluejoint	*Calamagrostis canadensis* (Michx. Beauv.	

[1] (Province of Alberta 2002)

Table 2.3. Mean soil, vegetation and leachate properties of uncontaminated, diesel fuel contaminated and crude oil contaminated soils

		Uncontaminated Disturbed Soil	Diesel Fuel Contaminated Soil	Crude Oil Contaminated Soil
Total Extractable Hydrocarbons[1]	2003	96 (33) b	12500 (645) a	10075 (950) a
	2004	34 (13) b	5213 (220) a	5863 (597) a
F2 Hydrocarbons[1] (>C11-C16)	2003	5 (0) c	6888 (537) a	2730 (297) b
	2004	5 (0) b	2064 (267) a	501 (51) b
F3 Hydrocarbons[1] (>C16-C34)	2003	63 (20) b	4698 (226) a	5278 (492) a
	2004	33 (11) b	3125 (186) a	3593 (355) a
F4 Hydrocarbons[1] (>C34-C60)	2003	34 (12) b	106 (19) b	1845 (96) a
	2004	6 (1) b	44 (26) b	1770 (270) a
Total Organic Carbon[2]	2003	2 (0.2) b	3.7 (0.3) a	3.5 (0.4) a
	2004	2.2 (0.3) b	3.3 (0.2) a	2.8 (0.2) b
Carbon to Nitrogen Ratio	2003	15.3 (0.9) b	16.2 (0.5) ab	17.8 (0.4) a
	2004	15.5 (1.5) b	15.2 (0.7) b	19.2 (1.1) a
Total Canopy Cover[2]	2003	71.6 (16.5) a	0.1 (0.1) b	0.8 (0.3) b
	2004	63.4 (2.8) a	5.4 (1.6) b	5.6 (2) b
Aboveground Biomass[3]	2004	499.28 (65.77) a	31.48 (4.71) b	22.58 (8.91) b
Belowground Biomass[3]	2004	75.1 (20.62) a	6.59 (2.3) b	6.53 (3.9) b
Leachate[4]	2004	44 (4) b	125 (18) a	125 (7) a

[1] units (mg kg^{-1}), [2] units (%), [3] units (g m^{-2}), [4] units (mm)
n = 4, numbers in brackets indicate standard error of the mean
Letters indicate significant differences ($p>0.05$, LSD) in a parameter between fuel types

Table 2.4. Mean soil property values in 2004 and mean change in soil property values between 2003 and 2004 (2004 minus 2003) in uncontaminated and contaminated soils

		Uncontaminated Soil				Diesel Fuel Contaminated Soil				Crude Oil Contaminated Soil			
		NA	F	C	CF	NA	F	C	CF	NA	F	C	CF
TOC	2004	2.2 (0.3) b	2.2 (0.2) b	6.7 (0.6) a	6.2 (0.4) a	3.3 (0.2) b	3.4 (0.2) b	6.7 (0.7) a	7 (0.3) a	2.8 (0.2) b	2.3 (0.1) b	6 (0.2) a	6 (0.3) a
	Δ	0.2 (0.2) a	0.7 (0.1) a	0.7 (0.7) a	-0.1 (0.2) a	-0.5 (0.2) ab	0.1 (0.4) bc	-1.6 (0.5) a	0.8 (0.4) c	-0.7 (0.4) a	-0.7 (0.4) a	-1.3 (0.7) a	-1.2 (0.4) a
TKN	2004	0.2 (0.03) b	0.2 (0.01) b	0.5 (0.04) a	0.5 (0.01) a	0.2 (0.01) b	0.3 (0.01) b	0.5 (0.02) a	0.5 (0.02) a	0.2 (0.02) b	0.2 (0.01) b	0.5 (0.01) a	0.4 (0.01) a
	Δ	0 (0.02) b	0 (0.01) ab	0 (0.07) b	-0.1 (0.04) a	0 (0.01) b	0 (0.03) b	-0.1 (0.03) a	-0.1 (0.06) ab	-0.1 (0.02) c	-0.1 (0.02) bc	-0.2 (0.02) ab	-0.2 (0.05) a
C:N	2004	15.5 (1.5) a	11.9 (1) b	13.4 (0.2) b	13.2 (0.6) b	15.2 (0.7) a	12.9 (0.1) b	13.2 (0.6) b	13.5 (0.7) b	19.2 (1.1) a	14 (0.5) b	13.8 (0.6) b	14.9 (1.1) b
	Δ	0.1 (0.7) a	4.4 (0.4) b	0.7 (0.4) a	1.4 (0.8) a	-1 (1) a	1.3 (0.4) ab	0.2 (0.7) ab	2.3 (1) b	1.4 (1) a	2.9 (0.6) a	1.2 (1) a	3.1 (1.2) a
NH_4^+	2004	3 (0.3) b	72.3 (33.5) a	4.5 (0.4) b	6.1 (0.5) b	4.3 (0.7) b	131.6 (49) a	5 (0.4) b	6.6 (0.9) b	2.4 (0.2) a	68.3 (26.7) a	3.5 (0.3) a	5.2 (0.6) a
	Δ	-3.5 (0.3) c	-739.6 (33.6) a	-7.8 (3.2) c	-609.6 (1.9) b	0.8 (0.7) d	-684.5 (52.9) a	-3.2 (1.1) c	-604.6 (1.1) b	-1 (0.2) c	-740.7 (26.7) a	-2.8 (0.4) c	-601 (0.5) b
NO_3^-	2004	3.2 (0.5) b	81.5 (18.5) a	3.2 (0.3) b	128.6 (29.9) a	1.8 (0.2) b	55.7 (3.9) a	3.4 (0.4) b	49.6 (18.3) a	1.5 (0.2) b	79.7 (8.1) a	2.2 (0.3) b	4.6 (2.3) b
	Δ	-5.1 (0.3) b	73.9 (18.1) b	-306.6 (50.9) a	-232.2 (28.8) a	-7.2 (0.5) bc	46.4 (3.8) c	-720.1 (299.7) a	-275.7 (35.6) b	-11.9 (7.5) b	76.1 (7.9) c	-336.6 (12.9) a	-282.7 (8.7) a
pH	2004	7.15 (0.05) b	5.11 (0.04) c	7.71 (0.01) a	7.3 (0.04) b	7.01 (0.04) a	5.26 (0.13) b	7.34 (0.06) a	7.4 (0.05) a	7.19 (0.04) b	5.23 (0.03) c	7.58 (0.05) a	7.25 (0.02) b
	Δ	0.5 (0) c	-1.51 (0.07) a	0.56 (0.06) c	0.07 (0.04) b	0.34 (0.09) b	-1.39 (0.12) a	0.09 (0.06) b	0.25 (0.02) b	0.26 (0.04) c	-1.7 (0.06) a	0.4 (0.04) d	-0.02 (0.08) b
EC	2004	0.54 (0.03) c	4.36 (0.37) b	4.39 (0.38) b	6.81 (1.01) a	0.48 (0.09) c	3.5 (0.49) b	5.29 (0.51) a	5.78 (0.23) a	0.54 (0.08) b	4.04 (0.12) a	4.15 (0.55) a	4.54 (0.18) a
	Δ	0.09 (0.03) c	3.88 (0.37) d	-5.63 (0.37) a	-2.97 (1.17) b	0.03 (0.18) c	2.77 (0.63) d	-6.16 (0.87) a	-3.67 (0.21) b	0.03 (0.09) b	3.36 (0.24) c	-5.88 (0.7) a	-6.02 (0.52) a
SAR	2004	0.4 (0.08) c	0.14 (0.04) c	6.75 (0.34) a	4.84 (0.51) b	0.68 (0.09) c	0.2 (0.04) c	6.46 (0.36) a	5.31 (0.48) b	0.69 (0.09) c	0.2 (0.04) c	5.98 (0.64) a	3.69 (0.38) b
	Δ	0.13 (0.08) c	-0.16 (0.04) c	-2.8 (0.69) b	-4.84 (0.72) a	0.33 (0.1) b	-0.4 (0.25) b	-3.34 (0.74) a	-3.29 (0.49) a	0.41 (0.09) c	-0.13 (0.04) c	-2.68 (0.7) b	-5.56 (0.34) a
Na	2004	14.9 (3.1) b	15.6 (4.9) b	565 (65.9) a	643 (129) a	22.5 (2.7) b	21.1 (6) b	677 (81.4) a	644 (66.6) a	25 (3.9) b	23.8 (3.8) b	548 (76.9) a	418 (43.1) a
	Δ	4.1 (3.1) b	4.9 (5) b	-807 (109) a	-735 (177) a	9.8 (4.1) b	-11.4 (21.2) b	-735 (162) a	-549 (91.1) a	14.3 (4.1) c	9.8 (3.8) c	-660 (83.4) b	-895 (59.5) a

n = 4, numbers in brackets indicate standard error of the mean

Δ = mean change in soil property values between 2003 and 2004

Letters indicate significant differences ($p>0.05$, LSD) in a soil parameter between amendments within a fuel type

TOC, total organic carbon (%); TKN, total kjeldahl nitrogen (%), C:N, carbon to nitrogen ratio; NH_4^+, ammonium (mg kg^{-1}); NO_3^-, nitrate (mg kg^{-1}); EC, electrical conductivity (dS m^{-1}); SAR, sodium adsorption ratio; Na, sodium (mg L^{-1}); Cd, cadmium (mg kg^{-1}); Cu, copper (mg kg^{-1}); Pb, lead (mg kg^{-1}); Zn, zinc (mg kg^{-1}); D_b, bulk density (Mg m^{-3})

Table 2.4. continued. Mean soil property values in 2004 and the mean change in soil property values between 2003 and 2004 (2004 minus 2003) in uncontaminated and contaminated soils

		Uncontaminated Soil				Diesel Fuel Contaminated Soil				Crude Oil Contaminated Soil			
		NA	F	C	CF	NA	F	C	CF	NA	F	C	CF
Cd	2004	0.5	0.5	1.25	1.16	0.5	0.5	1.29	1.19	0.5	0.5	0.89	1
		(0) b	(0) b	(0.10) a	(0.03) a	(0) b	(0) b	(0.14) a	(0.06) a	(0) b	(0) b	(0.09) a	(0.09) a
	Δ	0	0	0.2	-0.01	0	0	0.16	0.21	0	0	-0.06	-0.03
		(0) a	(0) a	(0.25) a	(0.07) a	(0) a	(0) a	(0.13) a	(0.05) a	(0) a	(0) a	(0.07) a	(0.19) a
Cu	2004	13.6	11.8	86.3	77.8	15.4	14.4	72.5	72.6	11.8	10.9	60.9	64.5
		(1) b	(0.9) b	(5.1) a	(1.9) a	(1.9) b	(0.5) b	(3.7) a	(4.7) a	(1.9) b	(0.7) b	(4.9) a	(6.2) a
	Δ	1.4	0	21	9	4.1	1.6	-1	-21.6	2.5	1.1	3.6	5.3
		(2.2) a	(0.8) a	(14) a	(2.6) a	(1.8) a	(0.3) a	(6.4) a	(40.7) a	(1.8) a	(0.7) a	(3.6) a	(9.9) a
Pb	2004	12.4	10.3	64.8	57	15.4	12.5	53.9	53.8	11	9.9	44.6	49.8
		(0.8) b	(0.8) b	(6.2) a	(2.5) a	(3.6) b	(0.7) b	(2) a	(1.2) a	(2.4) b	(0.5) b	(4) a	(4.8) a
	Δ	1.6	0	13	5	4.1	0.8	-2.6	9.5	3.3	0.6	1.9	5.8
		(1.9) a	(0.8) a	(10.3) a	(3.4) a	(3.7) a	(0.8) a	(4.2) a	(1.7) a	(2.1) a	(0.9) a	(3.8) a	(6.9) a
Zn	2004	54.5	46.3	270.8	233.5	55.8	54.5	233.1	225.1	45.8	49.8	185.8	198
		(4.1) c	(2.1) c	(11.2) a	(6.2) b	(2.7) b	(1.9) b	(10) a	(11.3) a	(5.7) b	(2.4) b	(14.8) a	(18.5) a
	Δ	2.5	-5.3	54	0.8	5.8	-0.8	-0.9	37.1	6.3	1	-4.3	0.5
		(6.9) a	(1.8) a	(23.7) a	(9.5) a	(3) a	(2.2) a	(14.4) a	(12.6) a	(6.3) a	(2.9) a	(11.7) a	(32.3) a
D_b	2004	1.33	1.3	1.23	1.07	1.2	1.15	1.05	1.03	1.29	1.27	1.08	1.09
		(0.03) a	(0.02) a	(0.05) a	(0.05) b	(0.04) a	(0.05) a	(0.03) b	(0.03) b	(0.03) a	(0.01) a	(0.03) b	(0.06) b

n = 4, numbers in brackets indicate standard error of the mean

Δ = mean change in soil property values between 2003 and 2004

Letters indicate significant differences ($p>0.05$, LSD) in a soil parameter between amendments within a fuel type

TOC, total organic carbon (%); TKN, total kjeldahl nitrogen (%), C:N, carbon to nitrogen ratio; NH_4^+, ammonium (mg kg^{-1}); NO_3^-, nitrate (mg kg^{-1}); EC, electrical conductivity (dS m^{-1}); SAR, sodium adsorption ratio; Na, sodium (mg L^{-1}); Cd, cadmium (mg kg^{-1}); Cu, copper (mg kg^{-1}); Pb, lead (mg kg^{-1}); Zn, zinc (mg kg^{-1}); D_b, bulk density (Mg m^{-3})

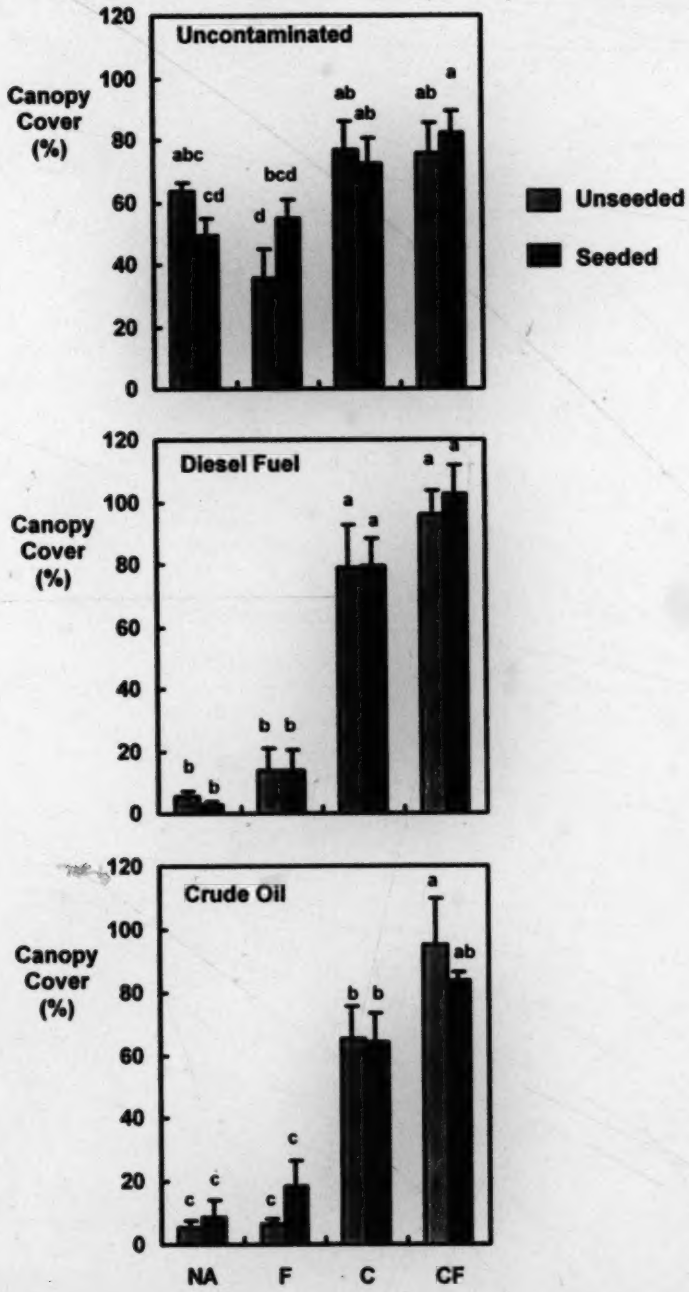

Figure 2.1. Total canopy cover on uncontaminated and contaminated soils in 2004. Letters indicate significant differences ($p<0.05$, LSD) among amendment and seeding combinations within a fuel type. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

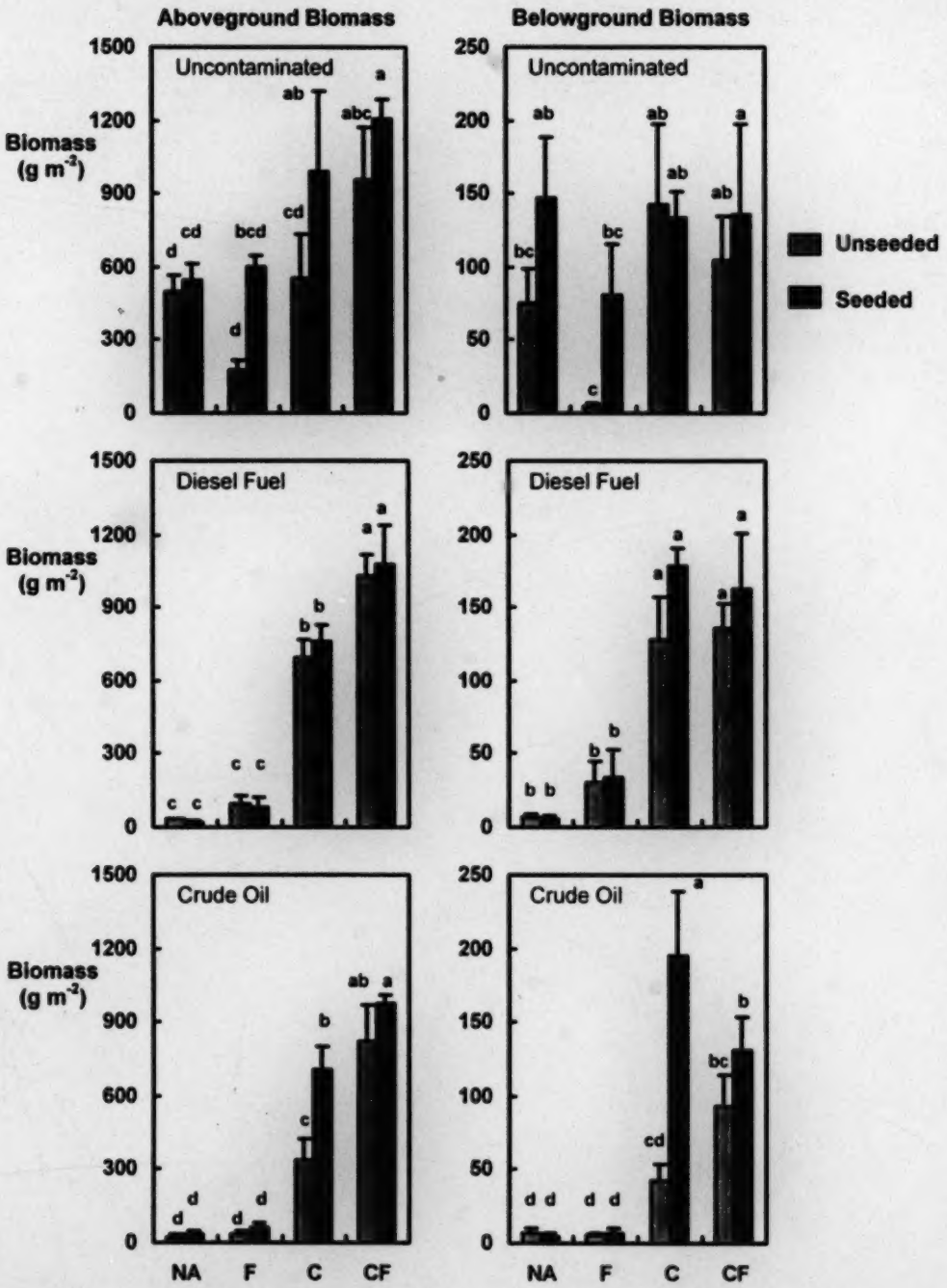

Figure 2.2. Above and belowground biomass from uncontaminated and contaminated soils in 2004. Letters indicate significant differences ($p<0.05$, LSD) in a parameter among amendment and seeding combinations within a fuel type. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer, seeded = seeded with native grass mix.

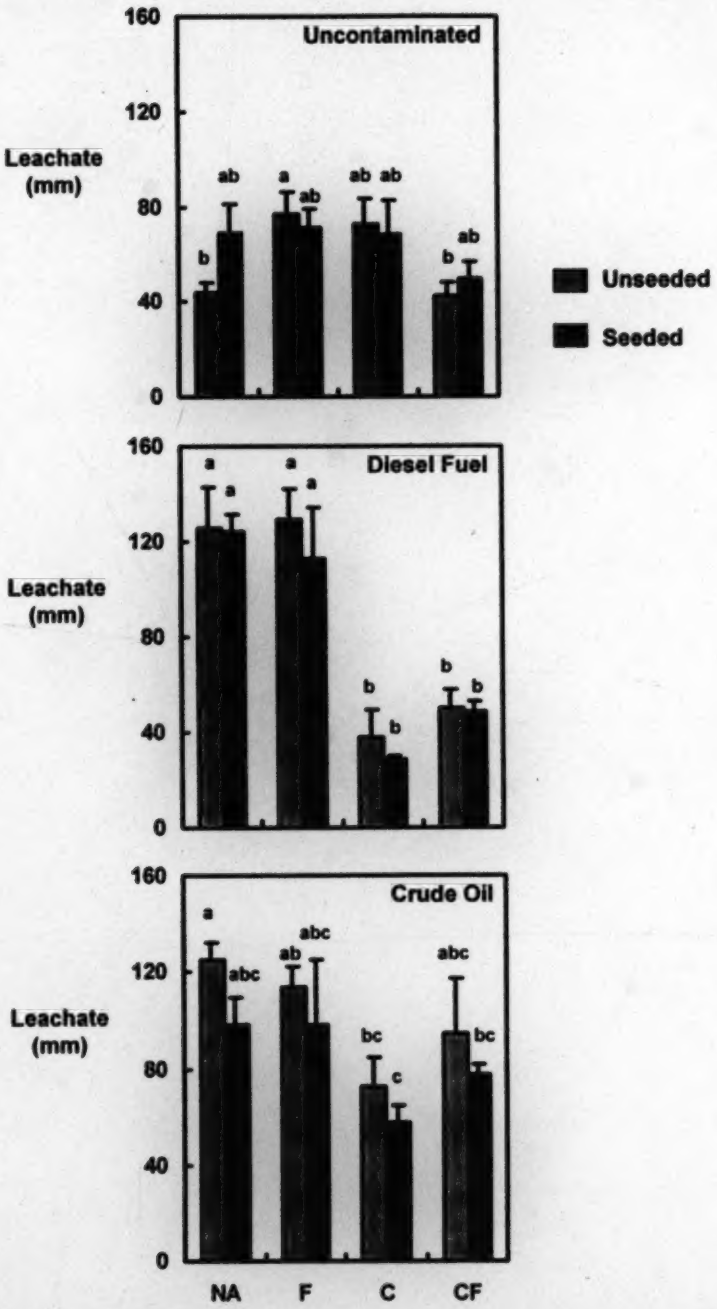

Figure 2.3. Cumulative leachate quantities from uncontaminated and contaminated soils. Letters indicate significant differences ($p<0.05$, LSD) among amendment and seeding combinations within a fuel type. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

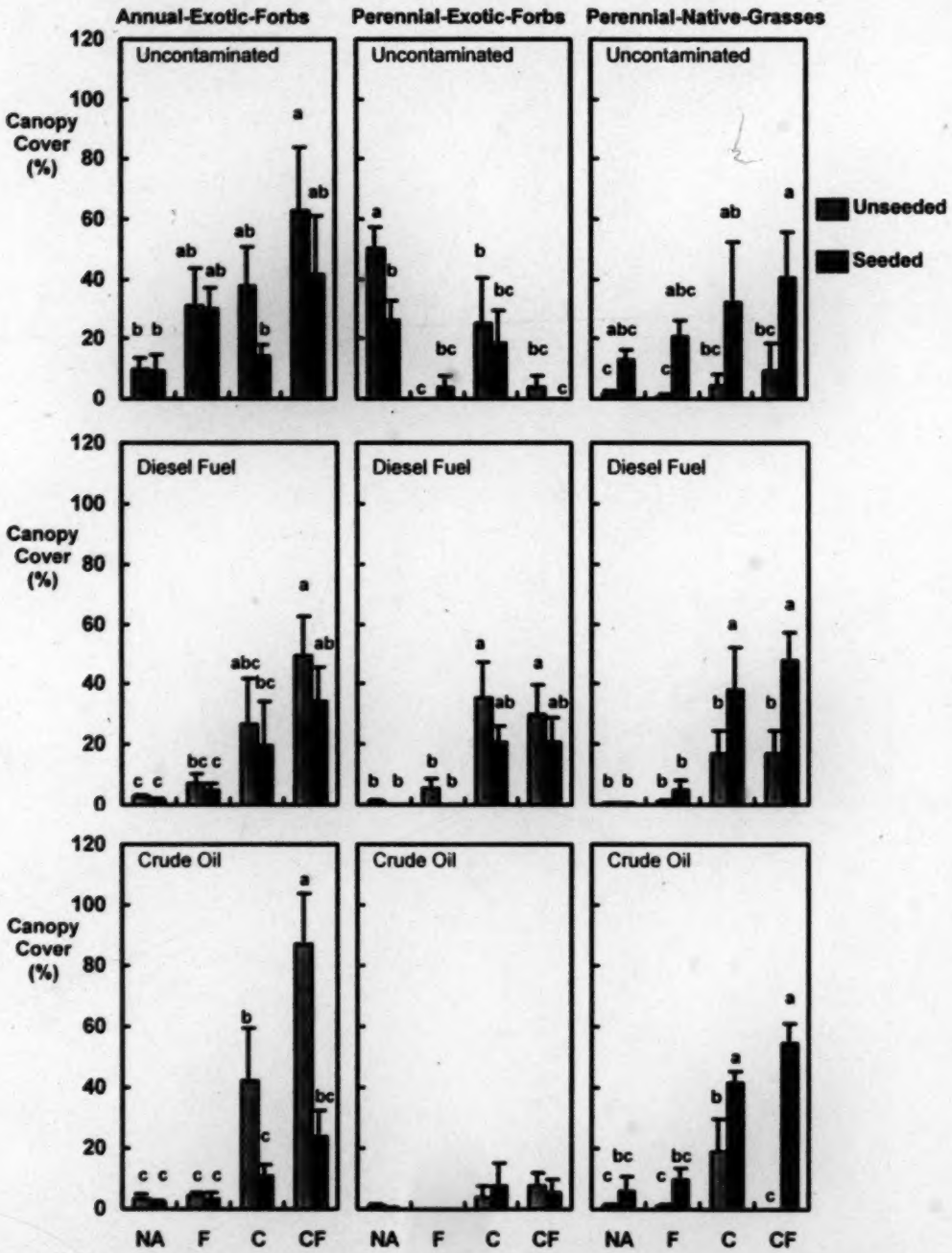

Figure 2.4. Canopy cover of annual-exotic-forbs, perennial-exotic-forbs and perennial-native-grasses on uncontaminated and contaminated soils in 2004. Letters indicate significant differences ($p<0.05$, LSD) in a parameter among amendment and seeding combinations within a fuel type. Absence of letters indicates no significant differences between treatments. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

3. BIOREMEDIATION OF HYDROCARBON CONTAMINATED SOILS

3.1 INTRODUCTION

Despite increased environmental awareness by the oil and gas industry, contamination of soils with petroleum hydrocarbons remains a substantial problem. Most petroleum hydrocarbon contamination results from mechanical failures, leaks or spills associated with oil and gas exploration and development activities. Due to environmental and health concerns, contaminated sites must be reclaimed to equivalent land capability (Province of Alberta 1995). Bioremediation is an effective method to treat contaminated soils as it involves microbial degradation of the contaminant. Bioremediation can be achieved either ex situ or in situ. Ex situ bioremediation refers to the excavation of contaminated soils and treatment on or off site. Due to the need for extensive soil handling, ex situ bioremediation can be prohibitively expensive and environmentally invasive. In situ bioremediation involves on site treatment of unexcavated soil where indigenous microorganisms metabolize soil contaminants (Liu and Suflita 1993, Amellal et al. 2001).

The main issue with in situ bioremediation is the unknown time required to achieve minimum acceptable hydrocarbon concentrations. Organic and/or inorganic amendments applied to contaminated soils can accelerate microbial degradation of hydrocarbons (Hupe et al. 1996: Brook et al. 2001: Namkoong et al. 2002). Organic amendments are known for supplying nutrients in a slow release form due to slow C and N mineralization rates (Levi-Minzi et al. 1990, Iglesias-Jimenez and Alvarez 1993, Sikora and Yakovchenko 1996). Inorganic fertilizers release available nutrients over a short timeframe, which has a priming effect on microbial activity and plant growth (McIntyre 2002). The vast majority of bioremediation studies have tested the combined effects of organic and inorganic amendments on biodegradation (Cole et al. 1995, Chang and Weaver 1998, Giles et al. 2001, Vouillamoz and Milke 2001), however there is a need to differentiate the individual and combined effects of organic and inorganic amendments on bioremediation success.

Vegetation can have a positive effect on hydrocarbon degradation in contaminated soils due to improved aeration, distribution of organic matter throughout the soil profile and increased microbial activity in the rhizosphere (Aprill and Sims 1990, Shimp et al. 1993, Schnoor et al. 1995, Palmroth et al. 2002). To stimulate microbial activity and therefore degradation rates, it is advantageous to seed species that develop fibrous root systems and establish rapidly (Aprill and Sims 1990, Huang et al. 2004). The majority of bioremediation studies conducted test the performance of agronomic rather than native species on contaminated soils (Bailey and McGill 1999, Huang et al. 2004). Native species are adapted to environmental conditions, such as drought, nutrient levels and salt concentrations, and therefore may be a suitable option for revegetating contaminated soils.

A vast number of bioremediation studies have been conducted, however only a few have studied in situ biodegradation under field conditions (Dibble and Bartha 1979, Jobson et al. 1974, Bhupathiraju et al. 2002, McIntyre 2002). There is a need for field bioremediation research, as ambient moisture and temperature conditions play a crucial role in microbial activity and bioremediation success (Atlas 1981, Leahy and Colwell 1990, Huesemann 1994). Inherent spatial variability of hydrocarbon spills, however, limits experimental control and replication adequacy (Nedunuri et al. 2000). Therefore, a manipulated field experiment was conducted to evaluate treatment effects on bioremediation success.

3.2 Research Objectives

- Investigate the effect of amendments and native seed mixes on hydrocarbon biodegradation in diesel fuel and crude oil contaminated soils
- Evaluate the persistence of hydrocarbon fractions in the amendment and seeding treatments in diesel fuel and crude oil contaminated soils

3.3 Materials and Methods

3.3.1 Study Area

The field experiment was established at the City of Edmonton Waste Management Centre in Edmonton, Alberta. The experiment was conducted from late June 2003 to late August 2004. The area has a mean annual temperature of 3.1 °C, the mean monthly temperature varies between -16.6 and 22.2 °C and the mean annual precipitation is 466 mm (Environment Canada 2004). At the research site, temperature conditions over the study period were similar to climate normals for the area, however rainfall was generally below normal in both growing seasons with the exception of August 2003 and July 2004 when above average rainfall occurred (Table A.1). The soil on site was an industrially disturbed black chernozem with a sandy loam texture. Due to the disturbed nature of the soil, total organic carbon levels were low relative to black soils in the region (van Ryswyk et al. 1965, Anderson and Coleman 1985, McGill et al. 1988) and average background soil hydrocarbon levels were < 100 mg kg^{-1} (Table 2.1).

3.3.2 Experimental Design

To ensure experimental control, experimental units in the study consisted of 20 L (28.5 cm diameter, 40 cm depth) plastic pails with an internal drainage system (perforated PVC tube (1.9 cm diameter) and sterilized sand (5 cm depth)) to remove leachate periodically (Figure A.1). A randomized complete block design was used in the experiment with three factors applied within each of four replicate blocks (Figure A.2). The factors tested in the experiment were fuel type, amendment combination and seed mix. Soil used in the experiment was a mixture of approximately 40% topsoil and 60% subsoil (based on dry soil mass) to represent an admixed disturbed soil. Soils were artificially contaminated with fuel to ensure experimental control over the type and uniformity of contamination. The fuel and amendment combinations were incorporated into the soil using a cement mixer (one treatment replicate mixed at a time). Soils were settled into the pails by gently tapping the bottom of the pail on the ground and then pails were buried in the ground (to an approximate depth of 35 cm) to ensure the treatments would be exposed to ambient

environmental conditions over the study period. Pails were seeded with the specified seed mix after soil placement and the surface was lightly raked to a depth of 0.5 cm to enable incorporation and minimize loss of seed. Leachate was removed from the pails following rainfall events to ensure the soils drained naturally and were not artificially exposed to anaerobic conditions.

Soils were contaminated with either diesel fuel or light crude oil, as they represent common contaminants in soil systems; an uncontaminated control was also monitored. Diesel fuel is characterized as containing predominantly light hydrocarbon fractions and light crude oil is known to contain a mixture of light and heavy fractions. Diesel fuel was obtained from Hughes gasoline station and crude oil was supplied by the ESSO Strathcona refinery in Edmonton, Alberta. Fuel was applied to the soil at a rate of 1% based on dry soil mass (equivalent to 10,000 mg kg^{-1} total extractable hydrocarbons) as this was viewed as a moderate contamination level suitable for bioremediation.

Soils were amended with non-source separated municipal solid waste (MSW) and sewage biosolids compost from the City of Edmonton, ammonium sulphate (21-0-0-24) fertilizer and a compost-fertilizer blend; an unamended control was also monitored. To maintain equal volumes of soil mixture in each pail, the fertilizer and unamended pails contained 23.2 kg of soil (dry mass) and the compost and compost-fertilizer pails contained 14.9 kg of soil (dry mass). The mature compost from the City of Edmonton contained macro and micronutrients, had a neutral pH and was pathogen free (fecal coliforms and *Salmonella*). However, due to the feedstocks of the material, it contained elevated concentrations of salts and certain metals (Cd, Cu, Pb, Zn) (Table A.2). Thus, the material is a class B compost according to Canadian Council of Ministers of the Environment (CCME) guidelines for compost quality (CCME 1995). The compost material was mixed with the soil at a rate of 20% based on dry soil mass (equivalent to 10% based on wet soil volume) in both the compost and compost-fertilizer treatments. This application rate was selected as it increased the total organic carbon level in the treatment soil to background levels for the area. Fertilizer application was aimed at optimizing nutrient levels for microbial activity to ultimately stimulate biodegradation of hydrocarbons. A C:N ratio of 10:1 was

considered ideal for microbial populations (Huesemann 1994), therefore ammonium sulphate was incorporated into the soil at a rate to achieve a hydrocarbon-C:N ratio of ~10:1 (Table A.3). Due to the lower quantity of soil in the compost-fertilizer treatment, 60 g of fertilizer was added to the compost-fertilizer treatment compared to 96 g added to the fertilizer treatment. The resulting C:N ratios in the fertilizer and compost-fertilizer treatments were 11.9 and 11.1 in the diesel soil, 11.1 and 11.8 in the crude soil and 8.6 and 12.1 in the control soil, respectively.

Soils were seeded with a native grass mix which included *Agropyron trachycaulum* (Link) Malte (slender wheatgrass), *Elymus innovatus* Beal (hairy wildrye), *Deschampsia caespitosa* (L.) Beauv. (tufted hairgrass) and *Calamagrostis canadensis* (Michx.) Beauv. (bluejoint) (Table A.4). An unseeded control was also monitored. Germination tests were conducted prior to seeding to confirm the viability of the seeds. Native species were selected based on seed availability, fibrous root systems and relative tolerance to salt, hydrocarbons and drought. The seeds were applied to the treatment pails at a rate of 300 pure live seeds m^{-2}, as this is a common rate used in reclamation.

3.3.3 Sampling

Soil samples were collected at the beginning and end of the study in June 2003 and August 2004, respectively. Samples were analyzed for total extractable hydrocarbons (TEH) between C_{11} and C_{60} (EPA 3550/8000-GC-FID), trace metals (EPA 6020), fecal coliform and *Salmonella* (APHA 2001), total organic carbon (TOC) (Nelson and Sommers 1996), total kjeldahl nitrogen (TKN) (McGill and Figueiredo 1993), ammonium (NH_4^+) (Maynard and Kalra 1993), nitrate (NO_3^-) (APHA 2001), sulphate (2004 only) (Combs et al. 1998), pH (Janzen 1993), electrical conductivity (EC) (Janzen 1993), sodium adsorption ratio (SAR) (Janzen 1993) and bulk density (2004 only) (ASTM 2004) (Table A.5). Individual carbon lengths were summed from the TEH analysis into three groups to correspond with fraction 2 (F2) (>C10 to C16), fraction 3 (F3) (>C16 to C34) and fraction 4 (F4) (>C34) hydrocarbons (CCME 2001). In addition,

three composite samples of topsoil, subsoil and compost were collected to characterize the initial materials in 2003. Samples were analyzed for above parameters in addition to bulk density and particle size distribution (Sheldrick and Wang 1993) (excluding compost samples). All samples were stored in coolers (~5 °C) during sampling and sent to EnviroTest Laboratories in Edmonton, Alberta for analysis immediately following collection.

In June 2003, one sample was taken from the cement mixer after the soil-fuel-amendment mixture for each treatment replicate had been homogenized. In August 2004, soil samples were collected from two depths in each pail, 0 to 10 cm and 10 to 20 cm, to evaluate differences in the soil profile over the study period. To collect the samples, each pail was removed from the ground, tipped upside down and then cut in half vertically to expose the center of the soil profile. Profile sampling was chosen as the soil was quite rocky and therefore soil coring was not effective. In addition, profile sampling enabled half the pail to be sampled for chemical analysis and the other half to be saved for root biomass collection. The cut edge was lightly scraped to remove potential contamination during cutting and samples were collected using a trowel from the two depths. All sampling equipment was sterilized periodically using a dilute bleach mixture to eliminate cross contamination between treatments. Upon analysis of the 2004 soils data, no statistically or biologically significant differences between the two sampling depths were found for all soil chemical parameters analyzed. Therefore, the depths were averaged for all parameters and the results presented reflect data from 0 to 20 cm.

Destructive vegetation sampling occurred at the end of the experiment in August 2004. Aboveground biomass (live vegetation and litter) was clipped from the entire pail area ($0.064\ m^{-2}$) prior to soil sampling and belowground biomass was collected via root washing. To obtain root biomass samples, an intact half pail of soil was placed on a 0.25 cm^2 wire screen and gently shaken. The soil was then sprayed with water to separate the soil particles from the root material. To ensure all the soil and organic material had been removed from the roots, the roots were gently massaged under water until only root material remained. All above and belowground biomass samples were oven dried at 80°C

for 48 hours then weighed. Both above and belowground biomass data were expressed on a mass per area basis (g m^{-2}) where the total rooting depth was 25 cm.

3.3.4 Data Analyses

Normality tests were conducted on the data prior to analysis and the majority of the parameters had normal distributions within a treatment type and the remaining data deviated only slightly from a normal distribution. Therefore, a two-way analysis of variance (ANOVA) with blocking was conducted using the amendment and seeding treatments as main factors. Interactions between these treatments were tested and multiple comparisons were conducted using Fisher's least significant difference (LSD) test (Zar 1999). Each fuel type was analyzed individually as the relationship between contaminated soils was not an objective of the study. All statistical were conducted using SYSTAT 10 (Systat Software Inc., Richmond, CA) software and a p-value of 0.05 was selected to determine significance.

3.4 Results

3.4.1 Contamination Effect

Contamination of soils with either diesel fuel or crude oil significantly increased the TEH concentration of the soil relative to the uncontaminated control (Table 3.2). Due to the properties of diesel fuel and crude oil, the contaminated soils contained different concentrations of hydrocarbon fractions. Concentrations of F2 hydrocarbons were significantly greatest in the diesel soil, followed by the crude soil and lowest in the uncontaminated soil. F3 hydrocarbons were greater in the diesel and crude soils relative to the uncontaminated soil and F4 fractions were greatest in the crude soil.

3.4.2 Soil and Vegetation Properties

In all three fuel treatments, there was a significant amendment effect on the soil properties and a non-significant seeding effect after the 14 month study period (Table

3.3). In the uncontaminated and contaminated soils, TOC and TKN concentrations were significantly greater in 2004 in compost and compost-fertilizer treatments relative to fertilizer and control treatments, however the unamended treatment had the significantly greatest C:N ratio at the end of the study. Final NH_4^+ concentrations were significantly greatest in the fertilizer treatment (non-significant in the crude soil). Final concentrations of NO_3^- were significantly greater in the fertilizer and compost-fertilizer treatments, except in the crude soil where NO_3^- concentrations were greatest in the fertilizer treatment. Overall, the fertilizer treatment had a significantly lower pH in 2004 relative to the other amendments. Electrical conductivity was significantly lower in the control treatment and similar among the fertilizer, compost and compost-fertilizer treatments at the end of the study. SAR and concentrations of Cd, Cu, Pb and Zn were significantly greater in the compost and compost-fertilizer treatments compared to the fertilizer and control treatments. Bulk density was significantly lower in compost and compost-fertilizer treatments, except in the uncontaminated soil where only the compost-fertilizer treatment had a significantly lower bulk density.

Overall, above and belowground biomass was affected by both amendments and seeding (Figure 3.1). In the unseeded uncontaminated treatment, above and belowground biomass was lower in the fertilizer treatment relative to the control and aboveground biomass was significantly greater in the compost-fertilizer treatment relative to the control. In unseeded diesel and crude soils, the compost and compost-fertilizer treatments had significantly greater above and belowground biomass than fertilizer and control treatments, with the compost-fertilizer treatment having significantly greater aboveground biomass than the compost treatment. Seeding affected biomass in only the compost treatment. Aboveground biomass was significantly greater in the uncontaminated soil and both above and belowground biomass was significantly greater in the crude soil when the compost treatment was seeded. No differences between seeded and unseeded treatments were observed in the diesel soil.

3.4.3 Fate of Hydrocarbons

Overall, hydrocarbon concentrations were significantly affected by amendment type. No statistical evidence of a seeding or amendment-seeding interaction was found for final hydrocarbon concentrations or the magnitude of change in hydrocarbon levels over time.

In the uncontaminated soil, no significant change was observed in the hydrocarbon content of the control and fertilizer treatments (Figure 3.2). However, the addition of compost and compost-fertilizer to uncontaminated soil led to an increase in TEH concentration (due to feedstocks of compost and high carbon content of material) and changes were observed over the study period (Figure 3.2). There was no statistical difference in the magnitude of change in F2 and F3 hydrocarbons in the compost and compost-fertilizer treatments. However the compost-fertilizer treatment had a significantly greater magnitude of change in F4 hydrocarbons than the compost treatment. Final levels of extractable hydrocarbons did not differ between the compost and compost-fertilizer treatments, but were both significantly greater than the fertilizer and control treatments in TEH and F3 and F4 hydrocarbon concentrations. Final levels of F2 hydrocarbons were not significantly different among amendment treatments.

In the diesel soil, there were statistically similar magnitudes of change in TEH and hydrocarbon fractions among amendment treatments, however final levels of extractable hydrocarbons were lowest in compost and compost-fertilizer treatments (Figure 3.3). Final TEH, F2 and F3 hydrocarbons were significantly lower in compost and compost-fertilizer treatments relative to fertilizer and unamended treatments. However, this trend was reversed for F4 hydrocarbons where final levels were significantly lower in fertilizer and unamended treatments compared to compost and compost-fertilizer treatments.

In the crude soil, the greatest reduction in extractable hydrocarbons was found in the compost-fertilizer treatment and lowest concentrations of extractable hydrocarbons were observed in compost and compost-fertilizer treatments (Figure 3.4). The magnitude of change in TEH and F3 hydrocarbons was significantly greater in the compost-fertilizer

treatment relative to the fertilizer and control treatments, with the compost treatment being statistically similar to all treatments. F4 hydrocarbons had the greatest magnitude of change in compost and compost-fertilizer treatments compared to fertilizer and control treatments. This trend was reversed for F2 hydrocarbons, with the greatest reduction over time found in the control. The final levels of all extractable hydrocarbons were significantly lowest in compost and compost-fertilizer treatments, followed by the fertilizer treatment and highest in the control treatment, with the exception of F4 hydrocarbons where all amendment types had significantly lower levels relative to the control.

3.5 Discussion

Due to the processing involved for petroleum fuels, the composition of hydrocarbon compounds varies greatly amongst fuels and also in soils contaminated with different petroleum hydrocarbons. Diesel fuels are characterized as containing shorter chain hydrocarbons in the range of C8 to C26, and comprised mainly of alkanes, both straight chained and branched, and aromatic compounds including mono, di and polyaromatic hydrocarbons (Adam and Duncan 1999, Gallego et al. 2001). On the other hand, light crude oil is known to contain longer chain hydrocarbons with more complex structures.

On contaminated sites, natural attenuation represents the 'do nothing' option in bioremediation. Inherent abiotic and biotic soil processes are capable of physical loss/immobilization or microbiological removal of hydrocarbons. Abiotic processes include volatilization, leaching and sorption to soil particles (Riser-Roberts 1998, Seklemova et al. 2001), whereas biotic processes involve degradation of petroleum hydrocarbon compounds by indigenous microorganisms (Liu and Suflita 1993). The decline in hydrocarbon content in the unamended treatments over the study period showed that the soil was capable of hydrocarbon removal or degradation. The coarse textured soil used in the study likely assisted natural attenuation as degradation of polycyclic aromatic hydrocarbons (PAH) has been greater in sandy soils relative to fine silt and clay textured soils (Amellal et al. 2001). However, the low vegetation biomass in

the unamended contaminated treatments suggests soil properties were limiting biological organisms and therefore likely hydrocarbon degradation.

Application of inorganic fertilizers to contaminated soils is among the most common technique studied in bioremediation research (Jobson et al. 1974, Dibble and Bartha 1979, Song et al. 1990, Carmichael and Pfaender 1997, Demque et al. 1997, Lawlor et al. 1997, Seklemova et al. 2001, McIntyre 2002, Atagana et al. 2003). During bioremediation, microorganisms incorporate carbon from petroleum hydrocarbons and inorganic nutrients into their cell structures (Huesemann 1994). However, contaminated soils commonly have imbalanced nutrient ratios due to the large input of carbon into the soil from the fuel (Atlas 1981, Leahy and Colwell 1990, Duncan et al. 1999, Atagana et al. 2003). Immobilization of available nutrients occurs when carbon to nutrient ratios increase, therefore inorganic fertilizers are applied to contaminated soils to supply readily available nutrients to the microbial populations.

The majority of studies conducted with inorganic fertilizers report enhanced hydrocarbon degradation (Jobson 1974, Demque et al. 1997, McIntyre 2002, Atagana et al. 2003). However, contradictory results were found in this study. The acidic pH of the fertilizer treatment can likely explain the limited stimulatory effect on degradation, as the optimal pH range for bioremediation is between 6 and 8 (Huesemann 1994, Alexander 1999). Microbial communities shift under acidic conditions and significant increases in degradation were found when acidic soils were neutralized (Leahy and Colwell 1990). Carmichael and Pfaender (1997) and Seklemova et al. (2001) observed little mineralization of phenanthrene and diesel fuel following fertilizer application with the hypothesis that inorganic nutrients stimulated microbial metabolism of readily degradable carbon compounds rather than metabolism of the contaminant.

Other soil properties such as water holding capacity, organic nutrients and organic matter content can limit degradation of hydrocarbons (Chang and Weaver 1998). Application of compost to contaminated soils can ameliorate poor soil conditions that limit bioremediation (Namkoong et al. 2002, Stukenholtz et al. 2002). In addition, compost

application can improve vegetation establishment on contaminated soils (Cole et al. 1995), and our results support this observation. Vegetation is a crucial component in bioremediation as microbial populations are significantly greater in the plant root zone. Bacterial populations in the rhizosphere can be two to three orders of magnitude higher than in surrounding soils and can also have greater metabolic capabilities (Aprill and Sims 1990, Anderson et al. 1994, Adam and Duncan 1999, Alkorta and Garbisu 2001). Binet et al. (2000) found low concentrations of PAHs in roots, and root and shoot tissue of ryegrass, therefore suggesting the major part of PAH dissipation in the rhizosphere is due to biodegradation or biotransformation.

Due to improvement in soil properties and vegetation establishment following compost application, increased hydrocarbon degradation has been observed in compost amended soils (Kästner et al. 1995, Kästner et al. 1999). The results of this study support these observations for TEH, F2 and F3 hydrocarbons. Hydrocarbon degradation in uncontaminated soils amended with compost consisted mainly of F4 hydrocarbons. Therefore the lower concentrations of TEH, F2 and F3 hydrocarbons in the compost amended contaminated soils can be attributed to petroleum hydrocarbon removal rather than compost hydrocarbon removal. Research results are also similar if hydrocarbon concentrations are converted to a mass basis (based on soil mixture mass), in that the compost and compost-fertilizer treatments have the lowest masses of TEH, F2 and F3 hydrocarbons relative to the other treatments (Figures A.3, A.4, A.5). Kästner and Mahro (1996) concluded that the organic matter in compost was likely the most important factor in degradation as the microflora responsible for PAH degradation were adapted to humic substances in the organic matter.

Numerous studies have been conducted which investigate the effect of compost and fertilizer combined on hydrocarbon degradation (Cole et al. 1995, Chang and Weaver 1998, Giles et al. 2001, Vouillamoz and Milke 2001). However, after an extensive literature review no studies were found that tested the effects of compost and fertilizer individually on degradation versus the combined amendment effect on degradation. Without such an investigation, it is difficult to partition the effects of compost and

fertilizer on degradation compared to a potential additive effect of the combined amendments. In this study, the lack of significant differences between compost and compost-fertilizer treatments in final hydrocarbon concentrations or magnitude of degradation suggests supplementing compost with fertilizer may not be necessary to stimulate hydrocarbon degradation.

Due to the source products and heterogeneous nature of compost materials, criticism exists towards use of compost in bioremediation. In particular, the high carbon content of composts has been shown to act as a competing nutrient source for microorganisms and ultimately decreases hydrocarbon degradation (Chang et al. 2001, Fernández et al. 2001). However, composts cured for long periods of time have low concentrations of readily mineralizable carbon and are therefore not subject to rapid microbial degradation when applied to soils (Levi-Minzi et al. 1990, Iglesias-Jimenez and Alvarez 1993, Sikora and Yakovchenko 1996). Another common trait of composts, particularly municipal solid waste composts, is elevated salinity and presence of heavy metals. As a result, the EC, SAR and metal content of receiving soils become elevated following compost application and can harm microbial and vegetation communities (Díaz-Marcote and Polo 1996). However, in this study the lower concentrations of hydrocarbons and higher vegetation biomass observed in the compost treatments would suggest the compost did not act as a competing nutrient source or negatively affect the soil environment. Namkoong et al. (2002) also observed no negative effects of compost application to degradation of diesel contaminated soils.

Remediation guidelines in Canada are set by the CCME and are based on hydrocarbons fractions rather than total levels (Table A.6). This approach addresses the relative ecotoxicity of certain groups of hydrocarbons and remediation limits are set accordingly (CCME 2001). All amendments used in this study were unable to degrade certain hydrocarbon fractions in the contaminated soils below CCME guidelines, however the compost and compost-fertilizer treatments were most successful. F2 hydrocarbons were below criteria and F3 hydrocarbons were above criteria in the compost and compost-fertilizer treatments in the diesel and crude soils. Such findings are not surprising as the

order of susceptibility of hydrocarbons to microbial attack is ranked as follows by decreasing susceptibility: *n*-alkanes, branched alkanes, low-molecular-weight aromatics, cyclic aromatics (Atlas 1981, Leahy and Colwell 1990, Song et al. 1990, Geerdink et al. 1996).

Mature composts contain high concentrations of carbon and specifically highly decomposed complex molecules. Compost materials may also contain hydrocarbon compounds from feedstocks such as municipal waste or sewage sludge. Therefore, when compost amended soils are analyzed for TEH, the carbon compounds in the compost are detected and result in an elevated hydrocarbon concentration of the sample. This creates difficulty when analyzing hydrocarbon concentrations from compost amended contaminated soils, as partitioning compost carbon from petroleum carbon is essentially impossible. In this study, soil samples from compost amended uncontaminated soil were compared with samples from compost amended contaminated soil to enable an indirect measurement of compost carbon in the contaminated soils. However, if the compost TEH were 'removed' from the samples, final concentrations of F3 hydrocarbons in the compost treated contaminated soils would still exceed CCME guidelines but by a much lesser degree.

Overall, the compost and compost-fertilizer treatments had the lowest concentrations of hydrocarbons in the diesel and crude soils at the end of the study. However, in the diesel soil all treatments had the same magnitude of change in hydrocarbon concentrations over time. In the crude soil, only the compost-fertilizer treatment had a greater magnitude of change in hydrocarbon concentrations relative to the fertilizer and control treatments, with the compost treatment being similar to all treatments. Further investigation into the long-term effects of compost and compost-fertilizer application is necessary to conclusively state whether compost or compost-fertilizer is superior in enhancing biodegradation. The results of this study suggest negligible differences between the two compost treatments, since final hydrocarbons levels and magnitude of hydrocarbon change over time did not differ between them.

3.6 CONCLUSIONS

- Relative to the unamended treatment, the fertilizer treatment had no stimulatory effect on degradation in the diesel soil, but had a small yet significant effect in the crude soil. This finding can likely be linked to the acidic pH of the treatment and the lack of plant growth.
- The lowest concentrations of TEH, F2 and F3 hydrocarbons were in the compost and compost-fertilizer treatments for both the diesel and crude contaminated soils. This finding can be linked to the higher organic matter content, increased plant biomass and enhanced water retention in the compost amended treatments. However, the final F3 hydrocarbon concentrations were above CCME guidelines in both compost treatments at the end of the study.
- The compost and compost-fertilizer treatments had similar levels of degradation and final hydrocarbon concentrations, however a long term study would be required to conclusively determine if a difference exists between the treatments.
- The current analytical method for total extractable hydrocarbons is unable to partition hydrocarbon-carbon present in the compost from hydrocarbon-carbon present in crude or refined petroleum products. Thus, alternative commercial laboratory methods for hydrocarbon analysis must be developed.

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Table 3.1. Mean baseline compost and soil treatment characterization

	Compost[1]	Soil[2]	Soil + Fertilizer[2]	Soil + Compost[2]	Soil + Compost + Fertilizer[2]
Sand (%)[1]	-	73 (0.5)			
Silt (%)[1]	-	14 (0.9)			
Clay (%)[1]	-	13 (0.5)			
Bulk Density ($Mg\ m^{-3}$)[1]	0.53 (0.01)	1.88 (0.02)	1.88 (0.02)	1.74 (0.01)	1.74 (0.01)
TEH ($mg\ kg^{-1}$)	9033 (491)	76 (15)	55 (13)	2053 (292)	2058 (163)
TOC (%)	19.9 (0.6)	1.8 (0.1)	1.6 (0.2)	6.4 (0.3)	6.3 (0.1)
TKN (%)	1.85 (0.01)	0.13 (0.01)	0.2 (0.01)	0.52 (0.02)	0.54 (0.01)
C:N	11.2 (0.4)	14.5 (0.5)	8.6 (0.7)	13 (0.2)	12.1 (0.3)
NH_4^+ ($mg\ kg^{-1}$)	7.5 (0.2)	6.1 (0.3)	811.8 (0.2)	10.7 (1.5)	638.2 (22.9)
NO_3^- ($mg\ kg^{-1}$)	1867 (75)	8.5 (0.4)	8.3 (0.5)	344.4 (19.9)	317.5 (28.8)
pH	7.7 (0.03)	6.63 (0.02)	6.65 (0.04)	7.13 (0.04)	7.19 (0.02)
EC ($dS\ m^{-1}$)	16.7 (0.47)	0.49 (0.02)	0.51 (0.02)	10.18 (0.28)	9.89 (0.11)
SAR	15.9 (0.4)	0.29 (0.01)	0.29 (0.01)	9.69 (0.25)	9.5 (0.12)
Na ($mg\ L^{-1}$)	2387 (89)	11.3 (0.4)	11 (0.4)	1399.2 (40.1)	1355 (22.5)
Cd ($mg\ kg^{-1}$)	3.0 (0.1)	0.5 (0)	0.5 (0)	1.01 (0.09)	1.04 (0.06)
Cu ($mg\ kg^{-1}$)	200 (2.8)	12.8 (0.6)	11 (0.6)	62.6 (5.5)	63.1 (3)
Pb ($mg\ kg^{-1}$)	152 (2.9)	11.1 (0.7)	9.7 (0.5)	47.3 (4.2)	46.5 (2.6)
Zn ($mg\ kg^{-1}$)	646 (9.9)	54.2 (2.6)	47.9 (1.9)	204.8 (17.5)	212.3 (12.9)
Fecal Coliforms (MPNU g^{-1})	<3 (0)	<3 (0)	<3 (0)	<3 (0)	<3 (0)
Salmonella	Not isolated	Not isolated	Not isolated	Not isolated	Not isolated

[1] n = 3

[2] n = 12 (seeding treatments combined)

Numbers in brackets indicate standard error of the mean

TEH, total extractable hydrocarbons; TOC, total organic carbon; TKN, total kjeldahl nitrogen, C:N, carbon to nitrogen ratio; NH_4^+, ammonium; NO_3^-, nitrate; EC, electrical conductivity; SAR, sodium adsorption ratio; Na, sodium; Cd, cadmium; Cu, copper; Pb, lead; Zn, zinc

Table 3.2. Mean soil hydrocarbon concentrations (mg kg^{-1}) in the uncontaminated, diesel fuel contaminated and crude oil contaminated soils in 2003

	Uncontaminated Disturbed Soil	Diesel Fuel Contaminated Soil	Crude Oil Contaminated Soil
Total Extractable Hydrocarbons	96 (33) b	12500 (645) a	10075 (950) a
F2 Hydrocarbons (>C11-C16)	5 (0) c	6888 (537) a	2730 (297) b
F3 Hydrocarbons (>C16-C34)	63 (20) b	4698 (226) a	5278 (492) a
F4 Hydrocarbons (>C34-C60)	34 (12) b	106 (19) b	1845 (96) a

n = 4, numbers in brackets indicate standard error of the mean
Letters indicate significant differences ($p>0.05$, LSD) in a parameter between fuel types

Table 3.3. Mean soil properties in fuel and amendment treatments in 2004

	No Amendment	Fertilizer	Compost	Compost + Fertilizer
Uncontaminated Soil				
TOC (%)	2.2 (0.3) b	2.2 (0.2) b	6.7 (0.6) a	6.2 (0.4) a
TKN (%)	0.2 (0.03) b	0.2 (0.01) b	0.5 (0.04) a	0.5 (0.01) a
C:N	15.5 (1.5) a	11.9 (1) b	13.4 (0.2) b	13.2 (0.6) b
NH_4^+ (mg kg^{-1})	3 (0.3) b	72.3 (33.5) a	4.5 (0.4) b	6.1 (0.5) b
NO_3^- (mg kg^{-1})	3.2 (0.5) b	81.5 (18.5) a	3.2 (0.3) b	128.6 (29.9) a
pH	7.15 (0.05) b	5.11 (0.04) c	7.71 (0.01) a	7.3 (0.04) b
EC (dS m^{-1})	0.54 (0.03) c	4.36 (0.37) b	4.39 (0.38) b	6.81 (1.01) a
SAR	0.4 (0.08) c	0.14 (0.04) c	6.75 (0.34) a	4.84 (0.51) b
Cd (mg kg^{-1})	0.5 (0) b	0.5 (0) b	1.25 (0.10) a	1.16 (0.03) a
Cu (mg kg^{-1})	13.6 (1) b	11.8 (0.9) b	86.3 (5.1) a	77.8 (1.9) a
Pb (mg kg^{-1})	12.4 (0.8) b	10.3 (0.8) b	64.8 (6.2) a	57 (2.5) a
Zn (mg kg^{-1})	54.5 (4.1) c	46.3 (2.1) c	270.8 (11.2) a	233.5 (6.2) b
D_b (Mg m^{-3})	1.33 (0.04) a	1.3 (0.03) a	1.23 (0.05) a	1.08 (0.05) b
Diesel Fuel Contaminated Soil				
TOC (%)	3.3 (0.2) b	3.4 (0.2) b	6.7 (0.7) a	7 (0.3) a
TKN (%)	0.2 (0.01) b	0.3 (0.01) b	0.5 (0.02) a	0.5 (0.02) a
C:N	15.2 (0.7) a	12.9 (0.1) b	13.2 (0.6) b	13.5 (0.7) b
NH_4^+ (mg kg^{-1})	4.3 (0.7) b	131.6 (49) a	5 (0.4) b	6.6 (0.9) b
NO_3^- (mg kg^{-1})	1.8 (0.2) b	55.7 (3.9) a	3.4 (0.4) b	49.6 (18.3) a
pH	7.01 (0.04) a	5.26 (0.13) b	7.34 (0.06) a	7.4 (0.05) a
EC (dS m^{-1})	0.48 (0.09) c	3.5 (0.49) b	5.29 (0.51) a	5.78 (0.23) a
SAR	0.68 (0.09) c	0.2 (0.04) c	6.46 (0.36) a	5.31 (0.48) b
Cd (mg kg^{-1})	0.5 (0) b	0.5 (0) b	1.29 (0.14) a	1.19 (0.06) a
Cu (mg kg^{-1})	15.4 (1.9) b	14.4 (0.5) b	72.5 (3.7) a	72.6 (4.7) a
Pb (mg kg^{-1})	15.4 (3.6) b	12.5 (0.7) b	53.9 (2) a	53.8 (1.2) a
Zn (mg kg^{-1})	55.8 (2.7) b	54.5 (1.9) b	233.1 (10) a	225.1 (11.3) a
D_b (Mg m^{-3})	1.2 (0.04) a	1.15 (0.05) a	1.05 (0.04) b	1.03 (0.04) b
Crude Oil Contaminated Soil				
TOC (%)	2.8 (0.2) b	2.3 (0.1) b	6 (0.2) a	6 (0.3) a
TKN (%)	0.2 (0.02) b	0.2 (0.01) b	0.5 (0.01) a	0.4 (0.01) a
C:N	19.2 (1.1) a	14 (0.5) b	13.8 (0.6) b	14.9 (1.1) b
NH_4^+ (mg kg^{-1})	2.4 (0.2) a	68.3 (26.7) a	3.5 (0.3) a	5.2 (0.6) a
NO_3^- (mg kg^{-1})	1.5 (0.2) b	79.7 (8.1) a	2.2 (0.3) b	4.6 (2.3) b
pH	7.19 (0.04) b	5.23 (0.03) c	7.58 (0.05) a	7.25 (0.02) b
EC (dS m^{-1})	0.54 (0.08) b	4.04 (0.12) a	4.15 (0.55) a	4.54 (0.18) a
SAR	0.69 (0.09) c	0.2 (0.04) c	5.98 (0.64) a	3.69 (0.38) b
Cd (mg kg^{-1})	0.5 (0) b	0.5 (0) b	0.89 (0.09) a	1.0 (0.09) a
Cu (mg kg^{-1})	11.8 (1.9) b	10.9 (0.7) b	60.9 (4.9) a	64.5 (6.2) a
Pb (mg kg^{-1})	11 (2.4) b	9.9 (0.5) b	44.6 (4) a	49.8 (4.8) a
Zn (mg kg^{-1})	45.8 (5.7) b	49.8 (2.4) b	185.8 (14.8) a	198 (18.5) a
D_b (Mg m^{-3})	1.29 (0.03) a	1.27 (0.02) a	1.08 (0.03) b	1.09 (0.06) b

n = 4, numbers in brackets indicate standard error of the mean

Letters indicate significant differences ($p<0.05$, LSD) in a parameter among amendments within a fuel type

TOC, total organic carbon; TKN, total kjeldahl nitrogen, C:N, carbon to nitrogen ratio; NH_4, ammonium; NO_3, nitrate; EC, electrical conductivity; SAR, sodium adsorption ratio; Cd, cadmium; Cu, copper; Pb, lead; Zn, zinc; D_b, bulk density

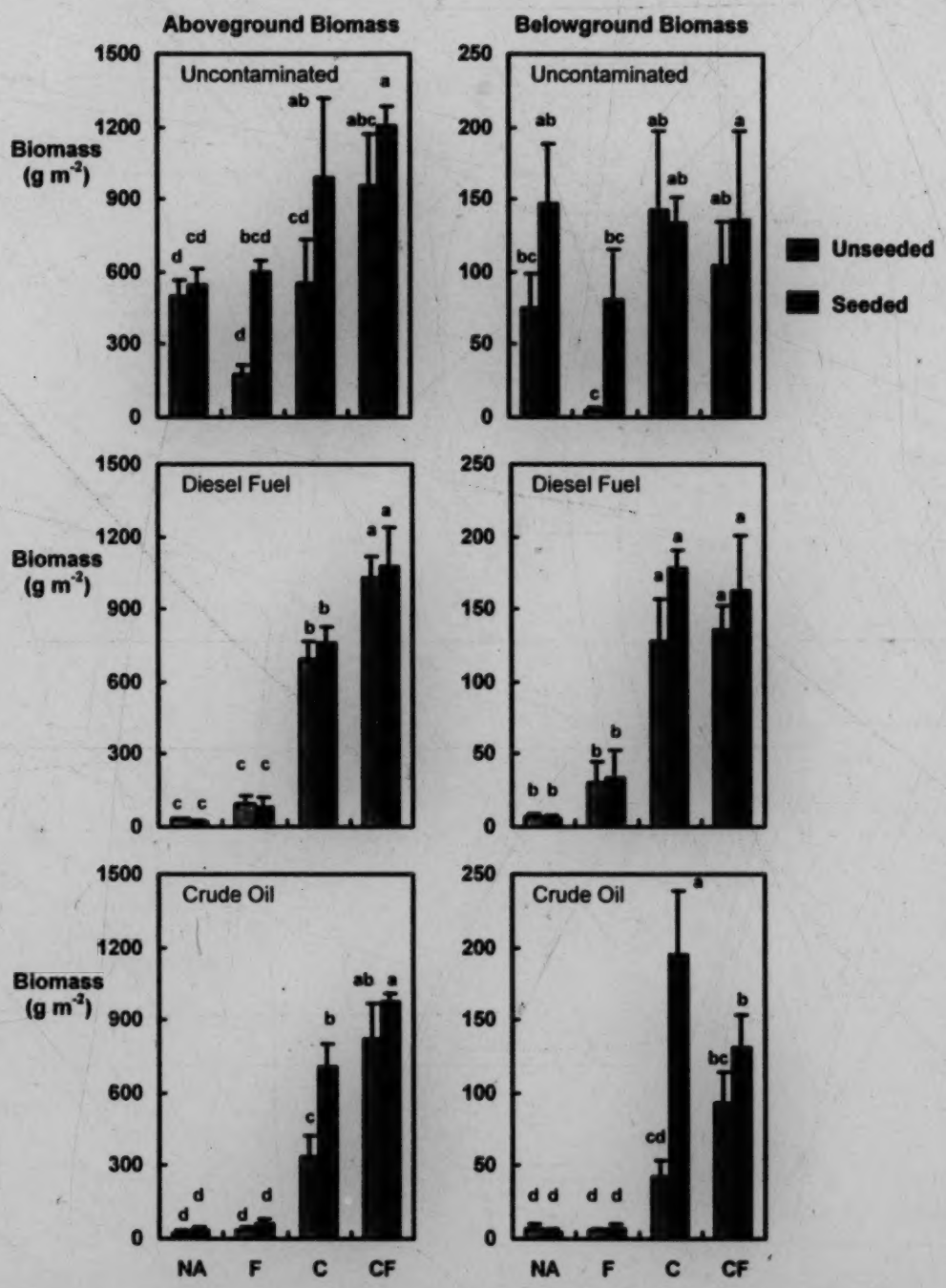

Figure 3.1. Above and belowground biomass from uncontaminated and contaminated soils in 2004. Letters indicate significant differences ($p<0.05$, LSD) in a parameter among amendment and seeding combinations within a fuel type. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

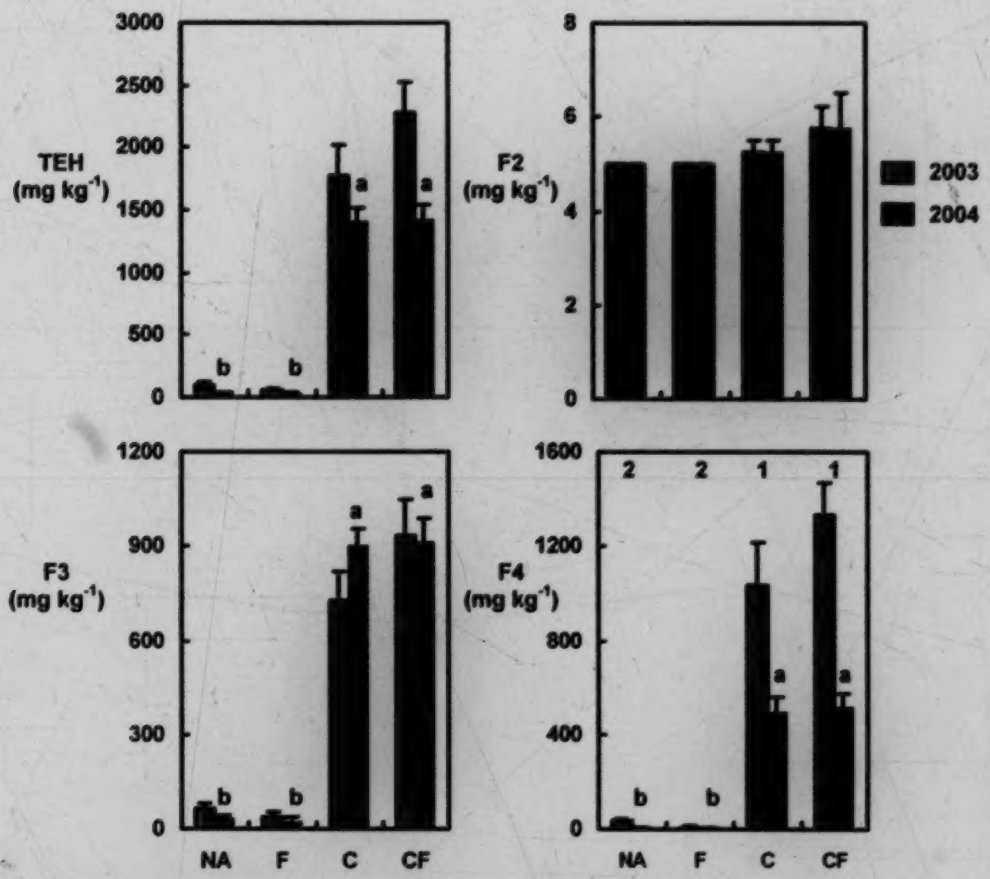

Figure 3.2. Mean extractable hydrocarbon concentrations in amended uncontaminated soils at the beginning (2003) and end (2004) of the study. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon concentrations among amendment types. Numbers indicate significant differences ($p<0.05$, LSD) in the magnitude of change in hydrocarbon concentrations between 2003 and 2004 among amendment types. Absence of letters or numbers indicates no significant differences between amendments. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

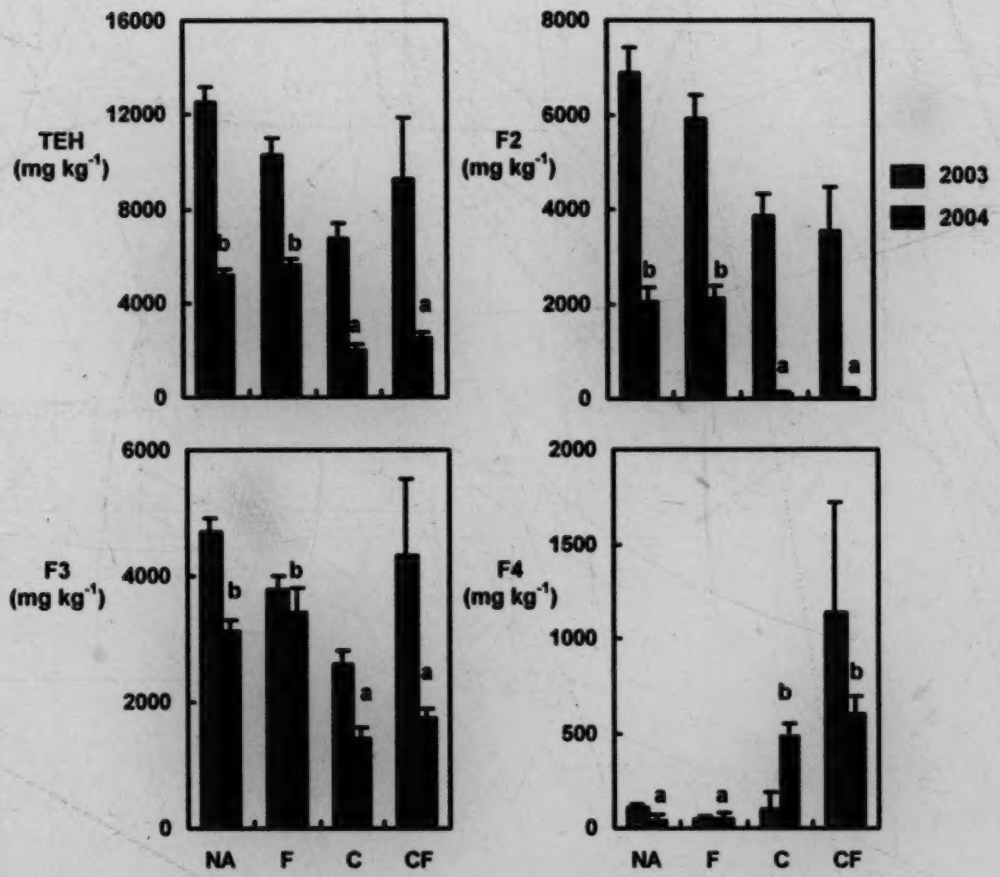

Figure 3.3. Mean extractable hydrocarbon concentrations in amended diesel fuel contaminated soils at the beginning (2003) and end (2004) of the study. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon concentrations among amendment types. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

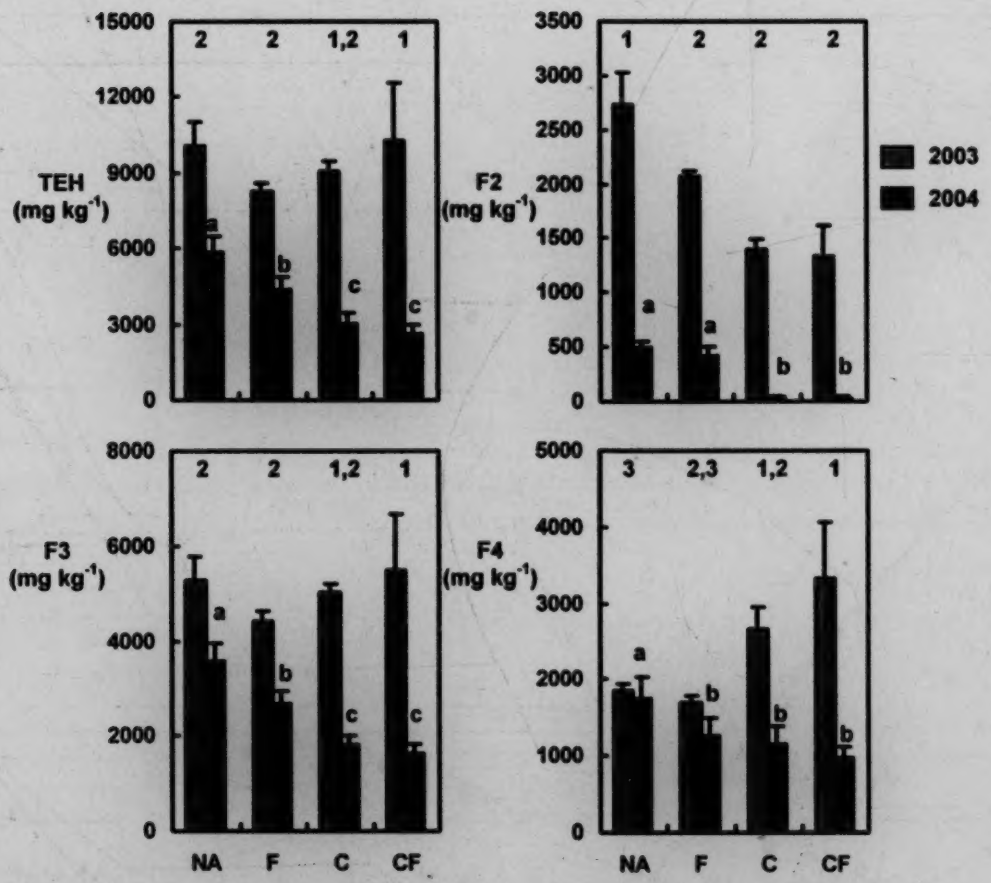

Figure 3.4. Mean extractable hydrocarbon concentrations in amended crude oil contaminated soils at the beginning (2003) and end (2004) of the study. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon concentrations among amendment types. Numbers indicate significant differences ($p<0.05$, LSD) in the magnitude of change in hydrocarbon concentrations between 2003 and 2004 among amendment types. NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix.

4. RECLAMATION OF HYDROCARBON CONTAMINATED SOILS: A SYNTHESIS

4.1 RESEARCH FINDINGS

Contamination with either diesel fuel or light crude oil increased the TEH concentration of the soil, which resulted in decreased vegetation cover and biomass and increased leachate production. The phytotoxic effects of fuel and potential water repellency are likely the main factors influencing vegetation growth and leachate on contaminated soils (Chaîneau et al. 1997). Hydrocarbon levels decreased in the control contaminated soils over the study, which proves the soils were capable of natural attenuation. However, hydrocarbon concentrations were greatest in the control soils relative to other treatments.

Addition of certain amendments to diesel fuel and light crude oil contaminated soils had a positive effect on hydrocarbon degradation and vegetation production. Compost amended treatments had the lowest hydrocarbon concentrations and highest cover and biomass of vegetation at the end of the study. The high organic matter content of compost likely reduced the phytotoxic effect of the contaminated soil (Chaîneau et al. 1997), facilitating vegetation establishment and therefore microbial stimulation in the rhizosphere (Aprill and Sims 1990). Fertilizer addition to contaminated soils resulted in little to no stimulatory effect on hydrocarbon degradation or vegetation production relative to the unamended treatment. The acidic pH of the fertilizer treatment is likely responsible for this relationship as microbial and vegetation populations shift under acidic conditions.

Seeding with native grasses had no effect on hydrocarbon degradation or total canopy cover, however the composition of vegetation was different between seeded and unseeded treatments. Perennial-native-grass cover was greater in all seeded treatments, suggesting such species were not present in the seed bank or seed rain. Cover of perennial-native-grasses was greatest in the compost and compost-fertilizer treatments in both the uncontaminated and contaminated soils.

Overall, compost amended treatments had the largest effect on hydrocarbon degradation and vegetation establishment. However, there was little difference between the compost and compost-fertilizer treatments, therefore fertilizer addition to compost may not be necessary for revegetation or bioremediation.

4.2 Future Research Directions

The stimulatory effect of compost on hydrocarbon degradation in soil profiles needs to be tested. In this study, there was no difference between soil parameters in the surface 10 cm and the 10 to 20 cm layer, however with increasing depths differences would likely be found. McIntyre (2002) found stimulatory degradation effects of inorganic fertilizer down to 70 cm, however no such studies have been conducted with compost materials. It is possible that compost-fertilizer treatments could achieve greater degradation rates deeper in the soil profile than compost treatments due to microbial stimulation outside the root zone from the fertilizer. However, this hypothesis needs to be tested.

Chemical analysis of leachate from contaminated soils amended with either compost or fertilizer should be conducted. Parameters that could be sampled include total extractable hydrocarbons, available nutrients and sodium. Understanding the mobility of these compounds is necessary to further evaluate environmental quality concerns associated with contaminated sites and treatment with amendments.

Degradation of hydrocarbons is typically rapid in the beginning when easily degradable compounds are mineralized and then slows with time as more complex recalcitrant compounds remain. Thus, there is a need for long term studies investigating degradation of F3 and F4 hydrocarbon compounds on in situ treatment sites.

4.3 Management Implications

During composting, decomposable organic materials are microbially converted to stable, pathogen and weed free organic compounds. Compost feedstocks are commonly waste

products such as manure, sewage sludge or municipal solid waste, therefore marketing can be challenging due to the public perception surrounding waste. The results of this research and others (Aprill and Sims 1990, Cuevas et al. 2000) support the use and positive effect of compost in land reclamation. Compost improves soil quality, promotes vegetation establishment and facilitates hydrocarbon degradation on contaminated lands. In addition, use of compost in reclamation creates a market and provides a productive end use for the material, which ultimately reduces landfill accumulations.

4.4 REFERENCES

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Chaîneau, C.H., J.L. Morel and J. Oudot. 1997. Phytotoxicity and plant uptake of fuel oil and hydrocarbons. J. Environ. Qual. 26:1478-1483.

Cuevas, G., R. Blazquez, F. Martinez and I. Walter. 2000. Composted MSW effects on soil properties and native vegetation in a degraded semiarid shrubland. Compost Sci. Util. 8:303-309.

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APPENDIX

Table A.1. Climate conditions at the research site relative to climate normals for the area

	Temperature (°C)		Rainfall (mm)		Snowfall (cm)
	Research Site	Climate Normals[1]	Research Site	Climate Normals[1]	Climate Normals[1]
2003					
June	14.8	15.5	55.9	87.1	0
July	18.2	17.5	91.7	91.7	0
August	17.5	16.6	116.6	68.9	0
September	11.0	11.3	9.1	42.3	1.5
October	6.5	5.6	10.7	10.5	7.8
November	-7.0	-4.1	1.3	1.9	17.9
December	-7.8	-9.6	0.0	0.8	22.3
2004					
January	-14.6	-11.7	0.0	1.3	24.5
February	*	-8.4	*	0.9	15.8
March	*	-2.6	*	2.1	16.8
April	*	5.5	*	13.1	13.4
May	9.5	11.7	20.3	45.1	3.5
June	15.2	15.5	19.3	87.1	0
July	17.4	17.5	129.5	91.7	0
August	15.1	16.6	35.3	68.9	0

[1] Canadian Climate Normals, 1971 to 2000, Edmonton City Centre Airport (Environment Canada 2004)

* Data not available

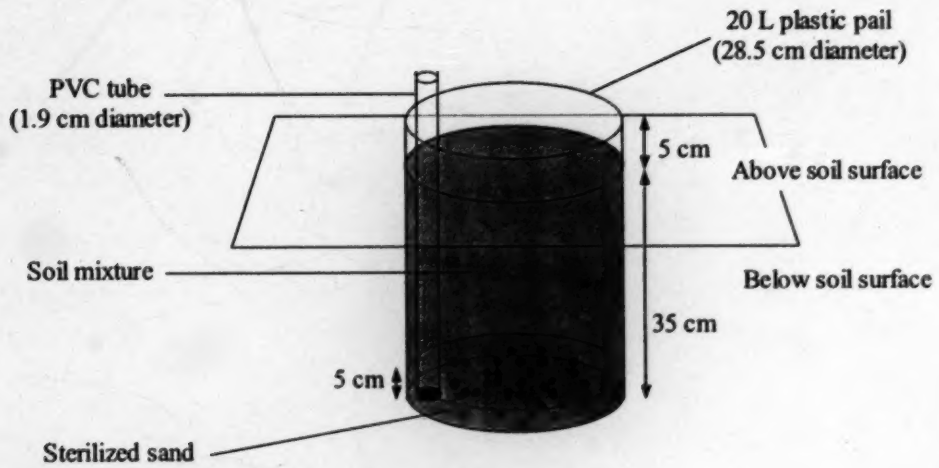

Figure A.1. Experimental units used in the field study.

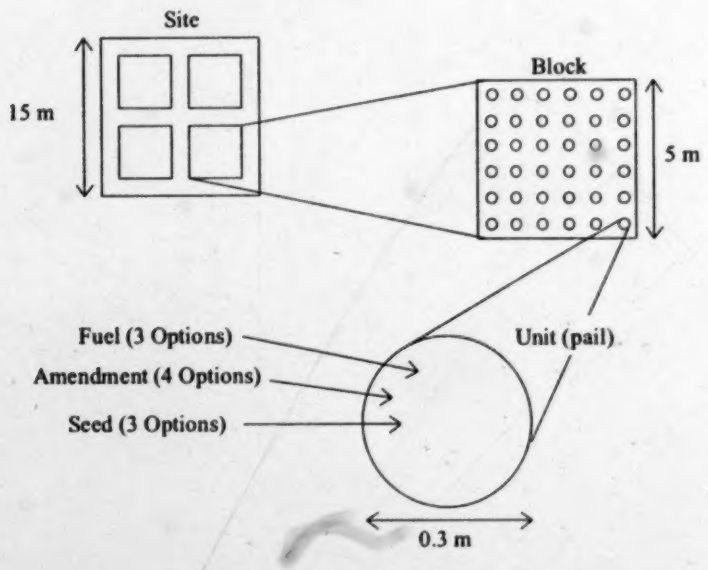

Figure A.2. Layout of experimental blocks and treatment pails at the study site.

Table A.2. Characterization of City of Edmonton municipal solid waste (MSW) and sewage biosolids compost relative to other MSW compost stocks and CCME guidelines

	City of Edmonton MSW-Biosolid Compost	MSW Compost Literature Values[2]	CCME Limits	
	Mean[1] (range)	(range)	Class A	Class B
Total Organic Carbon (%)	19.9 (19.2 - 21.2)	(12.4 – 43.2)		
Total Kjeldahl Nitrogen (%)	1.85 (1.84 - 1.86)	(0.45 – 3.1)		
Carbon to Nitrogen Ratio	11.2 (10.8 - 11.9)	(9.9 – 22)	< 25	
Moisture (%)	36.9 (35.4 – 37.8)	(25- 47.3)		
Nitrate (mg kg^{-1})	1866 (1730 - 1990)	16.5		
Ammonium (mg kg^{-1})	7.5 (7.1 - 7.7)	865		
pH	7.7 (7.6 - 7.7)	(6.3 – 8.9)		
Eelectrial Conductivity (dS m^{-1})	16.7 (15.8 - 17.3)	(2.4 – 16)		
Sodium Adsorption Ratio	15.9 (15.1 - 16.3)			
Sodium (mg L^{-1})	2387 (2210 - 2490)			
Fecal Coliforms (MPN g^{-1})	<3		<1000	
Salmonella (MPN $4g^{-1}$)	not detected		Non - detection	
Total Extractable Hydrocarbons (C_{11}-C_{60}) (mg kg^{-1})	9033 (8400 – 10000)			
Arsenic (mg kg^{-1})	5 (4.6 - 5.4)	(0.9 – 52)	13	75
Cadmium (mg kg^{-1})	3 (2.9 - 3.1)*	(0.04 – 100)	3	20
Chromium (mg kg^{-1})	81.7 (77.2 - 85.3)	(2 – 366)	210	1060
Cobalt (mg kg^{-1})	6.3 (6 - 7)	(15-48)	34	150
Copper (mg kg^{-1})	200 (194 - 203)*	(28 – 630)	100	757
Lead (mg kg^{-1})	152 (147 - 157)*	(9 – 900)	150	500
Mercury (mg kg^{-1})	0.65 (0.59 - 0.69)	(0.5 – 21)	1	5
Molybdenum (mg kg^{-1})	3.7 (3 - 4)	(1.0 – 25)	5	20
Nickel (mg kg^{-1})	53 (49 - 56)	(0.76 – 200)	65	180
Selenium (mg kg^{-1})	0.5 (0.4 - 0.6)	(0.5 – 3.6)	2	14
Zinc (mg kg^{-1})	646 (632 - 665)*	(37.6 – 1650)	500	1850

[1] n = 3

[2] (Gallardo-Lara and Nogales 1987, He et al. 1992, Iglesias-Jimenez and Alvarez 1993, Villar et al. 1993, He et al. 1995, Sikora and Yakovchenko 1996, Cuevas et al. 2000, Zhang et al. 2000, Marcote et al. 2002, Zaccheo et al. 2002)

* exceeds CCME class A guidelines

Table A.3. Calculation of ammonium sulphate (21-0-0-24) application rate

Fertilizer application rates were set to optimize the C:N between the hydrocarbon-carbon and the nitrogen in the fertilizer

Fertilizer Treatment

Soil = 25 kg
Fuel = 0.25 kg (contamination rate based on wet soil mass)

Carbon content in fuel = (0.25 kg)*(C content in fuel)
= (0.25 kg)*(12/14)
= 0.21 kg OR 210 g

To obtain C:N of 10, 21 g of nitrogen to be added to the soil

Amount of fertilizer required = (21 g)/(amount of N in fertilizer)
= (21 g)/(0.21)
= 100 g

However, in the experiment 96 g of fertilizer was added to the soil resulting in a C:N of 10.4 between the carbon in the fuel and the nitrogen in the fertilizer

Compost + Fertilizer Treatment

Soil = 16 kg
MSW = 5 kg
Fuel = 0.16 kg (contamination rate based on wet soil mass)

Carbon content in fuel = (0.16 kg)*(C content in fuel)
= (0.16 kg)*(12/14)
= 0.137 kg OR 137 g

To obtain C:N of 10, 13.7 g of nitrogen to be added to the soil

Amount of fertilizer required = (13.7 g)/(amount of N in fertilizer)
= (13.7 g)/(0.21)
= 65 g

However, in the experiment 60 g of fertilizer was added to the soil resulting in a C:N of 10.9 between the carbon in the fuel and the nitrogen in the fertilizer

Table A.4. Native grass species in the seed mixes and rationale for their selection

Species	Seed Mix		Selection Rationale[1]
	Native Grass	Native Grass + Legume	
Agropyron trachycaulum (Link) Malte (slender wheatgrass)	15%	10%	Short-lived, cool season, perennial, erect and tufted bunchgrass, fibrous root system, shade and salt tolerant, good establishment on oil sands tailings
Elymus innovatus Beal (hairy wildrye)	30%	20%	Perennial, sod forming grass, deep and spreading root system, tolerant of mildly saline soils, low drought tolerance, oil tolerance
Deschampsia caespitosa (L.) Beauv. (tufted hairgrass)	30%	20%	Cool season, perennial, densely tufted bunchgrass, medium salt tolerance, establishes on oil sands, moderately aggressive
Calamagrostis canadensis (Michx.) Beauv. (bluejoint)	25%	20%	Perennial, tufted grass, sometimes has short rhizomes, medium salt tolerance, moderately drought tolerant, good recovery noted on oil spill sites
Vicia americana Muhl. (american vetch)	-	15%	Perennial, N fixing, low drought and salt tolerance, no known hydrocarbon tolerance
Astragalus canadensis L. (Canada milk vetch)	-	15%	Perennial species with rhizomes

[1] (Hardy BBT Limited 1989)

Table A.5. Analytical methods used for soil and compost analysis

Parameter	Analytical Method	Source
Total Extractable Hydrocarbons (C11-60)	Gas chromatography – flame ionization detection	EPA 3550/8000-GC-FID
Trace Metals	Inductively coupled plasma – mass spectrometry	EPA 6020
Fecal Coliforms	Multiple tube fermentation technique (APHA 9221)	APHA 2001
Salmonella	Cultural isolation and serotyping (APHA 9260B)	APHA 2001
Total Organic Carbon	Combustion	Nelson and Sommers 1996
Total Kjeldahl Nitrogen	Titration	McGill and Figueiredo 1993
Available Ammonium	Extraction with KCl	Maynard and Kalra 1993
Available Nitrate	Colorimetry (APHA 4500)	APHA 2001
Available Sulphate	Turbidimetric procedure	Combs et al. 1998
pH	Saturation extract	Janzen 1993
Electrical Conductivity	Saturation extract	Janzen 1993
Sodium Adsorption Ratio	Calculation with $[Na^{+}]$, $[Ca^{+2}]$ and $[Mg^{+2}]$	Janzen 1993
Bulk Density	ASTM D5057	ASTM 2004
Particle Size	Hydrometer	Sheldrick and Wang 1993

Table A.6. Canadian environmental quality guidelines for soil

Parameter	Agricultural	Residential/ Parkland	Commercial	Industrial
F2 Hydrocarbons (>C11-C16) (mg kg^{-1}) [1]	450	150	760	760
F3 Hydrocarbons (>C16-C34) (mg kg^{-1}) [1]	400	400	1700	1700
F4 Hydrocarbons (>C34-C60) (mg kg^{-1}) [1]	2800	2800	3300	3300
Cadmium (mg kg^{-1}) [2]	1.4	10	22	22
Copper (mg kg^{-1}) [2]	63	63	91	91
Lead (mg kg^{-1}) [2]	70	140	260	600
Zinc (mg kg^{-1}) [2]	200	200	360	360
Electrical Conductivity (dS m^{-1}) [2]	2	2	4	4
pH [2]	6 to 8	6 to 8	6 to 8	6 to 8
Sodium Adsorption Ratio [2]	5	5	12	12

[1] (CCME 2001), for coarse textured soils (>75 μm)
[2] (CCME 2003)

Table A.7. Canopy cover (%) in August 2004 of vegetation in uncontaminated soil

Species List	NA		F		C		CF	
	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded
Setaria viridis L. (Beauv.)	0.5	0.5	8.3	5.3	1	0.1	0.1	0
Amaranthus retroflexus L.	0.3	1	1	5.4	4	0.5	0.9	0.3
Polygonum convolvulus L.	0	0.8	0.8	0	3.8	0	0	0
Chenopodium album L.	0	0	0.8	0	9.4	3.8	0	0
Kochia scoparia (L.) Schrad.	1	0.9	0.9	14	10.1	8.3	61.3	40.8
Crepis tectorum L.	7.8	5.4	9.6	1.8	7.6	1.1	0.3	0.3
Capsella bursa-pastoris (L.) Medic.	0	0.8	0	0	0.8	0	0	0
Portulaca oleracea L.	0.4	0.3	0.3	3.9	0.9	0.3	0	0.1
Polygonum aviculare L.	0	0	9.4	0	0	0	0	0
Artemisia absinthium L.	40.6	22.5	0	0	15.6	18.8	3.8	0
Taraxacum officinale Weber	9.4	3.8	0	3.8	9.4	0	0	0
Matricaria matricarioides (Less.) Porter	0.1	0.8	0	0	9.5	3.8	0	0
Erigeron canadensis L.	0	0	0	0	0	3.8	0	0
Brassica rapa L.	0	0	0	0	0	0	0	0
Echinochloa crusgalli (L.) Beauv.	0.8	0	3.8	0	0	0	0	0.1
Hordeum jubatum L.	1.8	5.4	0.8	13.3	4.5	4.5	9.4	0
Elymus innovatus Beal	0	0	0	0	0	3.8	0	0
Agropyron trachycaulum (Link) Malte	0	7.5	0	7.5	0	23.9	0	40.6
unidentified grass	0.9	0	0	0.1	0	0.1	0	0
unidentified forb	0	0	0	0.1	0	0	0	0.1
Total Canopy Cover	63.4	49.4	35.4	55	76.5	72.5	75.6	82.3

NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix

Table A.8. Canopy cover (%) in August 2004 of vegetation in diesel fuel contaminated soil

Species List	NA		F		C		CF	
	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded
Setaria viridis L. (Beauv.)	0	0.4	0.4	0.8	1.6	0.4	0.1	0.1
Amaranthus retroflexus L.	0.5	0.4	0.5	1.1	1	0.4	0.3	0.3
Polygonum convolvulus L.	0.3	0	0.9	0	0.1	0.8	0.8	0
Chenopodium album L.	0	0	0.1	0.8	9.4	0.9	0	3.8
Kochia scoparia (L.) Schrad.	0.1	0.3	0.1	0	0.9	16.5	44.4	20.1
Crepis tectorum L.	0.9	0.5	1.1	1.8	9.6	0.8	3.9	0.3
Capsella bursa-pastoris (L.) Medic.	0	0	0	0	3.9	0	0	0
Descurainia sophia (L.) Webb	0	0	0	0	0	0	0	9.4
Portulaca oleracea L.	1	0.3	4	0.8	0.1	0	0	0.1
Artemisia absinthium L.	0.9	0	0.8	0	35.1	20.6	28.1	16.9
Sonchus arvensis L.	0	0.1	0	0	0	0	0.8	0
Taraxacum officinale Weber	0	0	4.5	0	0	0	0.8	0
Plantago major L.	0	0	0	0	0	0	0	3.8
Matricaria matricarioides (Less.) Porter	0.3	0	0	0	0	0.8	0	0
Brassica rapa L.	0	0	0	0	0	0	0	0
Triticum aestivum L.	0	0	0	0	0	0	0	0
Echinochloa crusgalli (L.) Beauv.	0.9	0.1	0	3.8	0	0	0	0
Hordeum jubatum L.	0.4	0	1	0.9	16.9	13.1	16.9	18.8
Agropyron trachycaulum (Link) Malte	0	0.3	0	3.8	0	20.6	0	28.9
Deschampsia caespitosa (L.) Beauv.	0	0.1	0	0	0	4.5	0	0
Calamagrostis canadensis (Michx.) Beauv.	0	0.1	0	0	0	0	0	0
unidentified grass	0.1	0.3	0.3	0.1	0.3	0	0	0
unidentified forb	0.1	0.1	0	0.3	0	0	0.1	0
Total Canopy Cover	5.40	2.9	13.6	13.9	78.9	79.3	96.0	102

NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix

Table A.9. Canopy cover (%) in August 2004 of vegetation in crude oil contaminated soil

Species List	NA		F		C		CF	
	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded	Seeded
Setaria viridis L. (Beauv.)	0.3	0.1	1	1	9.8	0.3	0.8	0.3
Amaranthus retroflexus L.	0.4	0.5	1.1	0.5	1	0.4	3.9	1
Polygonum convolvulus L.	1.5	0.1	0.1	0.8	0.1	0.9	15.6	0
Chenopodium album L.	0	0	0	0	0	0.1	15.6	0.8
Kochia scoparia (L.) Schrad	0.1	0.4	0	0	4.5	5.3	32.8	16.9
Crepis tectorum L.	1.1	1.1	1	1.1	17	0.1	4	1
Capsella bursa-pastoris (L.) Medic.	0	0	0	0	9.5	0	13.9	3.8
Descurainia sophia (L.) Webb	0	0	0	0	0	3.8	0	0
Portulaca oleracea L.	0.1	0	1	0.1	0.4	0.3	0.8	0.3
Artemisia absinthium L.	0.8	0.1	0.1	0	3.8	3.8	3.8	4.5
Sonchus arvensis L.	0.1	0.1	0	0	0.1	0	0	0
Taraxacum officinale Weber	0	0	0	0	0	3.8	3.8	0.8
Androsace septentrionalis L.	0	0	0	0	0	0	0	0
Matricaria matricarioides (Less.) Porter	0.1	0.1	0	4.5	0	3.8	0	0
Potentilla norvegica L.	0	0	0.8	0	0	0	0	0
Brassica rapa L.	0	0	0	0	0	0	0	0
Echinochloa crusgalli (L.) Beauv.	0	0.1	0.1	0	0	0	0	0
Hordeum jubatum L.	0.9	1	0.9	0.8	18.8	9.5	0	4.5
Agropyron trachycaulum (Link) Malte	0	1	0	8.3	0	31.9	0	50
Deschampsia caespitosa (L.) Beauv.	0	4	0	0.8	0	0.1	0	0
unidentified grass	0	0	0.1	0.8	0.1	0.1	0.1	0
unidentified forb	0.3	0	0.1	0.1	0	0.3	0	0.1
Total Canopy Cover	5.6	8.8	6.4	18.6	65	64.1	94.9	83.8

NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix

Table A.10. Mean canopy cover (%) of annual-exotic-forb, perennial-exotic-forb and perennial-native-grass species in September 2003 and August 2004

Treatment		Annual Exotic Forbs		Perennial Exotic Forbs		Perennial Native Grasses	
		2003	2004	2003	2004	2003	2004
Uncontaminated Soil							
NA	Unseeded	62.6 (14.6)	9.9 (3.9)	8.3 (3.9)	50 (7.2)*	0 (0)	1.8 (0.7)
	Seeded	59.3 (10.7)*	9.5 (5.1)	2.3 (0.8)	26.3 (6.5)*	0.4 (0.2)	12.9 (3.4)*
F	Unseeded	0.9 (0.9)	30.9 (12.6)	0 (0)	0 (0)	0 (0)	0.8 (0.8)
	Seeded	20.9 (5.5)	30.3 (6.6)	0.8 (0.8)	3.8 (3.8)	1 (0.7)	20.8 (5.6)*
C	Unseeded	6.9 (3.7)	37.5 (13.1)	3.9 (3.7)	25 (15.3)	0 (0)	4.5 (3.6)
	Seeded	21.6 (14.4)	14 (4.2)	4.5 (3.6)	18.8 (10.8)	0.6 (0.1)	32.1 (20.2)
CF	Unseeded	1.8 (0.9)	62.5 (21.3)	0.1 (0.1)	3.8 (3.8)	0.3 (0.1)	9.4 (9.4)
	Seeded	9 (3.7)	41.4 (19.5)	0.1 (0.1)	0 (0)	4.5 (3.6)	40.6 (14.8)
Diesel Fuel Contaminated Soil							
NA	Unseeded	0 (0)	2.8 (0.8)*	0.1 (0.1)	0.9 (0.7)	0 (0)	0.4 (0.1)
	Seeded	0 (0)	1.8 (0.3)*	0.3 (0.1)	0.1 (0.1)	0.3 (0.1)	0.5 (0.2)
F	Unseeded	0.4 (0.2)	7.1 (3.2)	0.4 (0.2)	5.3 (3.5)	0 (0)	1 (0.7)
	Seeded	0 (0)	5.1 (1.8)	0 (0)	0 (0)	1 (0.7)	4.6 (3.5)
C	Unseeded	28.8 (16.7)	26.6 (14.9)	4.8 (3.5)	35.1 (12.2)	0.3 (0.1)	16.9 (7.7)
	Seeded	30.5 (10.8)	19.6 (14.7)	5.4 (3.3)	20.6 (5.6)*	1.1 (0.7)	38.3 (13.6)
CF	Unseeded	6.9 (2.8)	49.4 (12.9)*	6 (3.2)	29.6 (9.9)	0.3 (0.1)	16.9 (7.7)
	Seeded	16 (3.1)	34 (11.3)	1.1 (0.7)	20.6 (8.3)	2.1 (0.7)	47.6 (9.2)*
Crude Oil Contaminated Soil							
NA	Unseeded	0.1 (0.1)	3.5 (1.2)	0.3 (0.1)	0.9 (0.7)	0.3 (0.1)	0.9 (0.7)
	Seeded	0.1 (0.1)	2.3 (0.5)*	0 (0)	0.3 (0.1)	0.5 (0.2)	6 (5)
F	Unseeded	0 (0)	4.3 (1.3)*	0 (0)	0.1 (0.1)	0.1 (0.1)	0.9 (0.7)
	Seeded	0 (0)	3.5 (2)	0 (0)	0 (0)	0.8 (0.1)	9.8 (3.9)
C	Unseeded	9.4 (3.4)	42.3 (17.3)	0 (0)	3.9 (3.7)	0.1 (0.1)	18.8 (10.8)
	Seeded	11.1 (10)	11 (3.4)	0.1 (0.1)	7.5 (7.5)	1.1 (0.7)	41.5 (3.7)*
CF	Unseeded	7 (4.7)	87.3 (16.5)*	0.1 (0.1)	7.5 (4.3)	0 (0)	0 (0)
	Seeded	8 (3.5)	23.9 (8.6)	4 (3.8)	5.3 (4.3)	5.8 (3.3)	54.5 (6.3)*

n = 4, numbers in brackets indicate standard error of the mean

Asterisk indicates significantly ($p<0.05$, LSD) greater cover in a treatment between years within a vegetation class

NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer; seeded = seeded with native grass mix

Table A.11. Mean trace metal concentrations (mg kg^{-1}) in uncontaminated and contaminated soils in 2003 and 2004

		Uncontaminated Soil				Diesel Contaminated Soil				Crude Contaminated Soil			
		NA	F	C	CF	NA	F	C	CF	NA	F	C	CF
Arsenic	2003	6	6.4	5.8	7.1	5.8	6	5.8	5.8	6.3	5.8	6	5.3
	2004	6.3	6.4	6	6.8	6	5.6	6.4	9	6.7	6.3	7.1	6.2
Barium	2003	160	172	162	170	167	166	186	169	139	164	158	150
	2004	174	175	185	195	185	199	218	217	165	192	161	190
Beryllium	2003	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
	2004	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Cadmium	2003	<0.5	<0.5	1.1	1.2	<0.5	<0.5	1.1	1	<0.5	<0.5	1	1
	2004	<0.5	<0.5	1.3	1.2	<0.5	<0.5	1.3	1.2	<0.5	<0.5	0.9	1
Cobalt	2003	6	7	5	6	5	6	6	6	6	7	7	6
	2004	6	6	6	7	6	6	8	7	6	7	6	7
Chromium	2003	22.6	20.8	34.1	35.7	22.5	24.3	43	36.9	15.4	17.8	37.1	33.9
	2004	25.4	21.1	44.7	40.7	26.7	55.2	49.2	49.8	21.2	23.6	35.1	33.1
Copper	2003	12	12	65	69	11	13	74	94	9	10	57	59
	2004	14	12	86	78	15	14	73	73	12	11	61	65
Mercury	2003	<0.05	<0.05	0.24	0.25	<0.05	<0.05	0.27	0.23	<0.05	<0.05	0.28	0.28
	2004	0.06	0.06	0.23	0.18	0.07	0.05	0.21	0.18	0.05	0.06	0.15	0.15
Manganese	2003	943	943	748	703	768	718	778	760	998	895	713	705
	2004	876	921	923	783	928	799	838	906	928	771	678	750
Molybdenum	2003	<1	<1	1	1	<1	<1	2	1	<1	<1	1	1
	2004	<1	<1	1	1	<1	<1	1	2	1	<1	<1	<1
Nickel	2003	15	17	24	25	13	14	27	22	15	16	26	25
	2004	15	17	28	28	15	16	29	28	16	18	23	24
Lead	2003	11	10	52	52	11	12	57	44	8	9	43	44
	2004	12	10	65	57	15	13	54	54	11	10	45	50
Selenium	2003	0.4	0.4	0.3	0.4	0.4	0.4	0.4	0.4	0.3	0.4	0.6	0.6
	2004	0.4	0.4	0.5	0.5	0.3	0.3	0.5	0.5	0.7	0.4	0.4	0.5
Strontium	2003	19	20	52	57	21	22	55	45	17	19	46	47
	2004	22	19	57	54	21	22	50	52	20	20	46	47
Thallium	2003	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
	2004	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Vanadium	2003	18	19	15	16	17	17	16	16	18	20	17	16
	2004	22	21	19	20	22	22	22	22	22	25	18	19
Zinc	2003	52	52	217	233	50	55	234	188	40	49	190	198
	2004	55	46	271	234	56	55	233	225	46	50	186	198

NA = unamended; F = fertilizer; C = compost; CF = compost-fertilizer

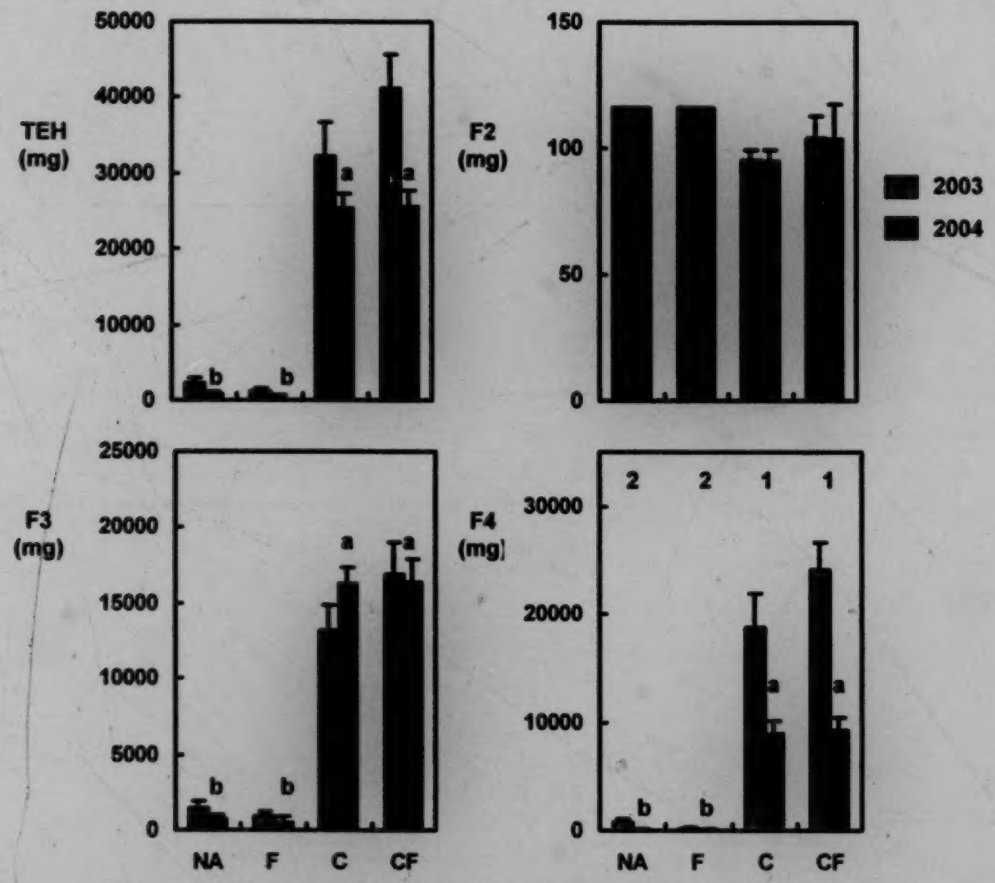

Figure A.3. Mean extractable hydrocarbon mass in amended uncontaminated soils at the beginning (2003) and end (2004) of the study. Hydrocarbon mass was calculated using total soil mixture mass for each treatment. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon mass among amendment types. Numbers indicate significant differences ($p<0.05$, LSD) in the magnitude of change in hydrocarbon masses between 2003 and 2004 among amendment types. Absence of letters or numbers indicates no significant differences between amendments.

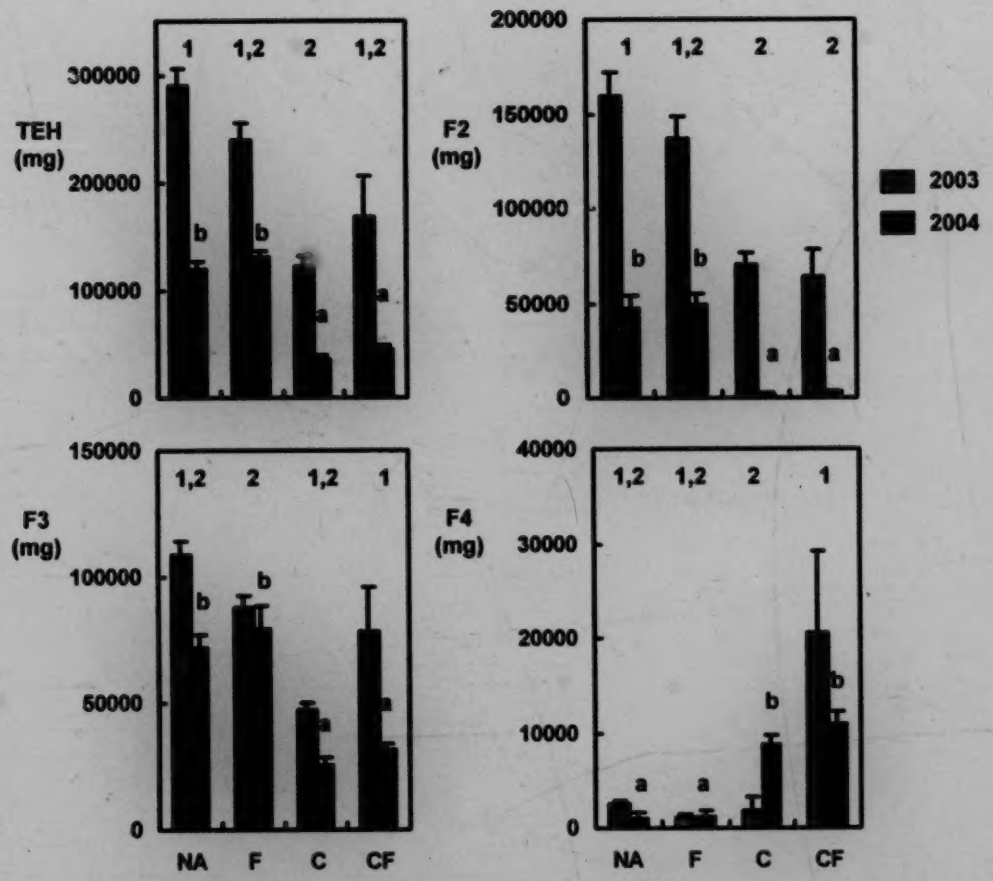

Figure A.4. Mean extractable hydrocarbon mass in amended diesel fuel contaminated soils at the beginning (2003) and end (2004) of the study. Hydrocarbon mass was calculated using total soil mixture mass for each treatment. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon mass among amendment types. Numbers indicate significant differences ($p<0.05$, LSD) in the magnitude of change in hydrocarbon masses between 2003 and 2004 among amendment types.

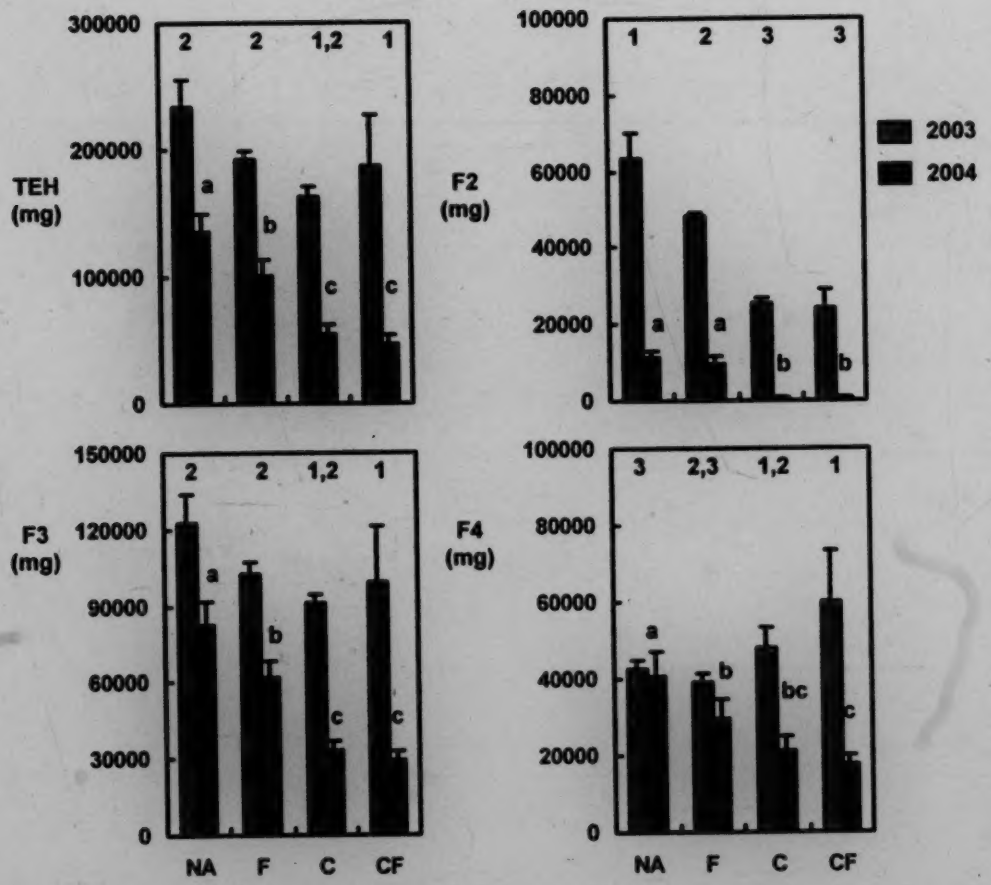

Figure A.5. Mean extractable hydrocarbon mass in amended crude oil contaminated soils at the beginning (2003) and end (2004) of the study. Hydrocarbon mass was calculated using total soil mixture mass for each treatment. Error bars indicate standard error of the mean. Letters indicate significant differences ($p<0.05$, LSD) in final hydrocarbon mass among amendment types. Numbers indicate significant differences ($p<0.05$, LSD) in the magnitude of change in hydrocarbon masses between 2003 and 2004 among amendment types.

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